Planning and Evaluating Restoration of Aquatic Habitats from an Ecological Perspective

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PREFACE

The work reported herein was conducted as part of the Evaluation of Environmental Investments Research Program (EEIRP). The EEIRP is sponsored by the Headquarters, U.S. Army Corps of Engineers (HQUSACE). It is jointly assigned to the U.S. Army Engineers Water Resources Support Center (WRSC), Institute for Water Resources (IWR) and the U.S. Army Engineer Waterways Experiment Station (WES), Environmental Laboratory (EL). Mr. William J. Hansen of IWR is the Program Manager and Mr. H. Roger Hamilton is the WES Manager. Technical Monitors during this study were Mr. John W. Bellinger and Mr. K. Brad Fowler, HQUSACE. The Field Review group members that provide overall Program direction and their District or Division affiliations are: Mr. David Carney, New Orleans; Mr. Larry M. Kilgo, Lower Mississippi Valley; Mr. Richard Gorton, Omaha; Mr. Bruce D. Carlson, St. Paul; Mr. Glendon L. Coffee, Mobile; Ms. Susan E. Durden, Savannah; Mr. Scott Miner, San Francisco; Mr. Robert F. Scott, Fort Worth; Mr. Clifford J. Kidd, Baltimore; and Mr. Edwin J. Woodruff, North Pacific.

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The report was compiled and edited by Dr. David J. Yozzo and Mr. John P. Titre, EL, and Ms. Jane Sexton, PTI Environmental Services, Bellevue, WA. Ms. Michaele Reynolds, PTI Environmental Services, Bellevue, WA, performed the final technical edit. Dr. Yozzo was supported by a National Research Council-WES Research Associateship. The report was prepared by PTI Environmental Services under Contract No. DACW39-93-0006 to Barry A. Vittor and Associates.

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At the time of publication of this report, Mr. Kyle E. Schilling was Acting Director, WRSC, and Dr. Robert W. Whalin was Director of WES. Commander of WES was COL Bruce K. Howard.

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ACRONYMS AND ABBREVIATIONS

CAD	confined aquatic disposal
CDE	coupled differential equation
Corps	U.S. Army Corps of Engineers
DAMOS	disposal area monitoring system
DOC	dissolved organic carbon
DOM	dissolved organic matter
EEIRP	Evaluation of Environmental Investments Research Program
EPA	U.S. Environmental Protection Agency
GIS	geographic information system
HEP	Habitat Evaluation Procedure
HGM	hydrogeomorphic
MLW	mean low water
PAH	polycyclic aromatic hydrocarbon
POC	particulate organic carbon
POM	particulate organic matter
PRT	Platte River Whooping Crane Maintenance Trust
PSWH	permanent shallow water habitat
SAV	submerged aquatic vegetation
USFWS	U.S. Fish and Wildlife Service
WET	Wetland Evaluation Technique

1. INTRODUCTION

Pace Wilber and John Titre

PURPOSE OF REPORT

The Civil Works Program of the U.S. Army Corps of Engineers (Corps) recognizes the importance of habitat restoration as an aspect of the federal government's responsibility to serve as an active steward of our nation's natural resources (e.g., Section 1135 of the Water Resources and Development Act of 1986, Section 204 of the Water Resources and Development Act of 1992). The Corps believes its role in meeting this responsibility will increase. In anticipation of that increase and to better meet existing responsibilities, the Evaluation of Environmental Investments Research Program (EEIRP) identified impediments to restoration planning within the Corps planning process and proposed alternative guidance. This report addresses these issues from an ecological perspective; other EEIRP reports directly address engineering, economic, planning, and institutional concerns.

This report describes important ecological processes and characteristics that should be considered when restoring aquatic and marine habitats. It is written for engineers, planners, and managers who do not have extensive backgrounds in ecology and for biologists new to habitat restoration. The purpose of the report is to foster discussions within multidisciplinary planning teams, not take the place of those discussions. Detailed, local knowledge and experience is the best source of information for planning habitat restoration projects, and teams of individuals that collectively have broad ranges of expertise are the best project planners (NRC 1992). Chapters 1 through 4 cover ecological issues relevant to most restoration projects, regardless of habitat type, and should be useful to all readers. Chapter 5 is divided into sections based on ecosystems; readers may want to focus only on the ecosystem-specific discussions relevant to their work.

SYNOPSIS OF REPORT

Chapter 1: Introduction. This chapter briefly overviews the report, describes how it was prepared, and summarizes three important concepts that emerged during the report's preparation. Four chapters follow these introductory comments.

Chapter 2: Ecological Planning Processes for Restoration Projects. This chapter describes the conceptual framework that this report is based on. It outlines a process for restoration planning that emphasizes concepts discussed in later chapters, such as adaptive management and ecosystem processes. It also discusses when planning requires the site-specific information outlined in Chapter 5. The fundamental message of Chapter 2 is that clear, technically sound, and ecologically-based objectives are essential to restoration planning. Once these goals are set, it is relatively easy to identify the environmental parameters and the temporal and spatial scales needed to achieve those goals. Clear goals will also make it easy to identify the parameters that measure the project's success or failure.

Chapter 3: Incorporating Ecological Theory into Restoration Project Planning. Discussions begun in Chapter 2 about links between ecological science and restoration planning are continued in this chapter. Chapter 2 refers to some of these links when outlining the planning processes that form the framework for this report. Chapter 3 explains these links in more detail and addresses links not covered in the previous chapter. The fundamental message of Chapter 3 is that ecosystems are dynamic and stochastic, not static and deterministic. For a project to meet its intended public uses (i.e., to be successful), it must persist or evolve in an acceptable manner. Interactions between organisms, such as succession and competition, and natural disturbance processes have the potential to modify a project's long-term outcome, and these modifications could diminish the intended uses of a project, perhaps to such a degree that the public views the project as a failure. Project designs must take these forces of change into account. This is relatively easy for physical processes because they are relatively well understood and modeled. Ecological processes are more problematic, but just as important.

Chapter 4: Goal Setting and Adaptive Management. Adaptive management is an effective way to manage restoration projects because it recognizes that long-term commitments are necessary to make most projects successful and that a variety of actions and adjustments may be necessary to achieve a desired outcome. After introducing the concepts of goal setting and adaptive management, this chapter provides a general model for the implementation of adaptive management that contrasts project objectives against performance in a defensible fashion. The model envisions a project progressing through a matrix of ecological states that are unique combinations of structural and functional characteristics specifically defined for that project. Progress through this matrix is not necessarily linear nor is it irreversible. Long-term, but not necessarily intensive, monitoring is needed to determine where a projects falls in the matrix and where it might be heading. Long-term commitments by responsible parties are needed to ensure appropriate actions are taken when necessary to redirect the project's evolution.

Chapter 5: Ecosystem and Restoration Profiles. This chapter takes the general concepts discussed in the previous chapters and illustrates their application in specific freshwater and coastal habitats. There are six ecosystem sections presented: 1) open coastline and near coastal waters, 2) subtidal estuaries, 3) estuarine and coastal wetlands, 4) freshwater wetlands, 5) streams and rivers, and 6) lakes and reservoirs. Adjacent terrestrial ecosystems are described only incidentally. Unfortunately, any divisions between habitat types in a report such as this will sacrifice ecological principles for matters of convenience. The divisions presented herein are good for site-level planning, which is the most common type of restoration project currently undertaken by the Corps and the type of project most Corps funding mechanisms target. However, these divisions do not work well for watershed-level planning because watersheds include multiple aquatic habitat types, as well as upland ecosystems. In this case, it may be necessary to read several ecosystem sections, as well as the materials dealing with the relevant terrestrial ecosystems, to obtain the desired information. Each section describes the ecosystem, key environmental processes, and the types of restoration projects that commonly occur within that ecosystem. The later subsection is problem oriented (i.e., it lists ecological problems that commonly occur in the habitat along with likely solutions to those problems). Two or more case studies are included in the *Restoration Projects* section for each ecosystem to illustrate the range of ongoing restoration activities both by project scope and geographic location. These studies were also chosen to identify the effectiveness of restoration planning and management in meeting project objectives and whether project implementation exhibited the following three criteria:

1. Adaptive management rather than traditional fixed-scope contracting procedures

2. Ecosystem-level (e.g., watershed) planning rather than species-specific restoration

3. Safe-fail approach to judging success, considering the tendency of systems to reach an ecologically valuable equilibrium even if it is not the intended outcome.

HOW THIS REPORT WAS PREPARED

Private consultants, academics, and Corps staff prepared this report during a 2-year period. Collectively, this group, identified in the Preface, has experience restoring the major aquatic and coastal habitat types found in the continental United States; several members have previously served on environmental review boards from the National Research Council, United Nations, and several states. The group met periodically to review progress and discuss emerging issues. These meetings usually involved visits to nearby restoration projects conducted by the Corps or state or local governmental agencies. These visits helped emphasize points made during meeting discussions and provided concrete examples of real-world problems and solutions.

SUMMARY OF KEY CONCEPTS

What is Restoration?

There are many definitions of habitat restoration, and it is easy for discussions to go awry because of these differences. The important point is to recognize that human desires, ecological history, and ecological and engineering feasibility all play roles in planning habitat restoration. These diverse inputs can lead to a variety of potential endpoints for a particular restoration project, and it should be recognized that more than one endpoint may be acceptable to the public. NRC (1992) distinguishes between three general restoration goals, which are useful when discussing habitat restoration: *restoration* returns an ecosystem to a close approximation of its condition before it was disturbed, *rehabilitation* improves a system to a "good working order," and *management* manipulates a system to ensure maintenance of one or a few functions. These concepts overlap and may be thought of as a continuum. Discriminating between the potential endpoints of a restoration project typically involves deciding which part of the restoration/rehabilitation/management continuum one is targeting. Throughout this report, the term *restoration* refers to this entire continuum.

Ecosystem Perspectives and Spatial Scales

It is important to understand and clearly articulate the spatial scale of habitat restoration projects because crossing scales confuse many issues and confounds communication. It also is important to view projects in a watershed or ecosystem context (U.S. GAO 1994; Interagency Ecosystem Management Task Force 1995). For some projects, the exact parcel of land that will be restored is known before restoration planning begins because it is the only land available for a project. In this situation, ecological and engineering constraints will likely yield few alternative endpoints for the project, and taking an ecosystem perspective will likely equate to considering how the project will be affected by processes originating outside its borders.

In some cases, the watershed or ecosystem in which the project will occur and the project budget are known before the planning process begins. In this situation, the objective becomes how to best make use of those funds in the system (i.e., what will give the most ecological improvement and what will win the most public support given the general location and funding constraints. Furthermore, the symptoms of environmental degradation that drew attention to the area may not be the best symptoms to address, so project goals should be flexible in the early planning stages. For example, complaints about channel-induced erosion to upland bird habitat may be why the Corps was asked to examine the feasibility of habitat restoration in an area, but additional marsh habitat may be what the ecosystem needs more. Finally, the objective (and budget) may be to restore an entire watershed or ecosystem. In this case, the questions becomes "What needs to be changed for the area to meet our intended uses and what is the best path to that endpoint?"

Adaptive Management

Chance events affect the results of restoration projects, and many project design decisions include considerable amounts of uncertainty. NRC (1992), U.S. GAO (1994), and Interagency Ecosystem Management Task Force (1995) recommend dealing with these realities by taking an adaptive management approach to habitat restoration. Adaptive management is an interactive process that regularly reexamines choices in the light of past outcomes. "Safe-fail" approaches (i.e., initially building several alternative designs that target the same endpoint so that subsequent actions can capitalize on best results) can be an important part of this implementation.

Adaptive management is relatively common for large projects, although project managers rarely use this term. Budget and logistical constraints typically require large restoration projects (e.g., the Kissimmee River restoration) to be implemented incrementally, allowing information about the results of early stages to be considered when designing and implementing later stages. The amount of information collected about the progress of early stages differs between projects, but most groups recognize the benefits of detailed information. Adaptive management is less common with smaller projects because institutional pressures encourage agencies to construct projects quickly and minimize post-implementation assessments. Instead, projects should be implemented in a tiered fashion that allows information about results from early tiers to be factored into the implementation of successive tiers.

Adaptive management requires flexible goals and designs and a long-term commitment to detailed monitoring and fine tuning after initial implementation. The potential benefits of adaptive management are great and widely recognized within the governmental community, so institutional impediments to its use should not persist.

REFERENCES

Interagency Ecosystem Management Task Force. 1995. The ecosystem approach: healthy ecosystems and sustainable economies, Volumes I and II.

NRC. 1992. Restoring aquatic ecosystems: science, technology, and public policy. National Research Council Commission on Geosciences, Environment, and Resources. National Academy Press, Washington, DC.

U.S. GAO. 1994. Ecosystem management: additional actions needed to adequately test a promising approach. GAO/RCED-94-111. U.S. General Accounting Office, Washington, DC.

2. ECOLOGICAL PLANNING PROCESS FOR RESTORATION PROJECTS

Robert Pastorok and Anne MacDonald

Effective planning is critical for restoration projects to maximize the overall success of restoration efforts and minimize costs. The planning process proposed in this chapter is based on experience with numerous restoration projects on a variety of scales. Many past projects embrace the key elements of the planning process but not necessarily all of the details described below. It has been found that many small to moderate-sized restoration projects have failed to achieve their potential because project planning efforts have been too limited. At best, projects in this category have succeeded on restricted terms (e.g., a good project for the site but not necessarily the optimum project for the region) and done as well as they have only because the professional judgment of the project planners was seasoned with years of accumulated local ecological knowledge. Commonly, the restoration objectives of such a project are vague, making success or failure impossible to evaluate. At worst, projects have failed to provide the minimum ecological functions necessary for the site because of an inadequate understanding of the ecological system being manipulated (e.g., a riverine backwater area fills with sediment or becomes anoxic within a decade of construction because watershed sediment routing, nutrient cycling, and hydrology were poorly understood).

Planning for any restoration project should define clear objectives that are quantitative statements of physical conditions (e.g., 20 percent attenuation of the 10-year flood peak) or biological results (e.g., the composition and structure of biological communities to be achieved) or a characterization of optimal habitats for target species. The success of the planning process depends on identifying key ecological processes within the ecosystem of concern and understanding those processes in relation to the objectives of the project. Identifying these processes and determining appropriate restoration objectives requires a relatively sophisticated understanding of the ecological principles and natural history described in later chapters of this report.

This chapter describes a useful ecological planning process for habitat restoration projects. Following an overview, the following major elements of the ecological planning process are addressed: 1) the importance of defining objectives; 2) the role of ecological models, restoration hypotheses, and key ecological parameters; 3) suggestions for dealing with uncertainty and avoiding the pitfalls inherent in many restoration projects; 4) restoration designs, feasibility, and experimentation; and 5) the stages of implementation and monitoring.

OVERVIEW OF THE PLANNING PROCESS

The primary steps in the ecological planning process proposed here are to:

- Define **habitat** of concern and existing **problem(s)**
- Develop **objectives** for restoration
- Develop a **conceptual model** of the ecosystem to be restored
- Develop **restoration hypotheses** (e.g., regarding responses to specific habitat manipulations or transplant efforts)
- Identify **key ecological parameters** to be manipulated or monitored
- **Evaluate restoration hypotheses** using **ecological models** or **reference site** information
- Develop restoration **design**
- Perform **feasibility, cost, and impact analysis**
- Develop final restoration design and **implementation plan**
- Perform monitoring and **adaptive management**.

Figure 2-1 shows the relationships among these steps in the ecological planning process and the supporting evaluations. Supporting steps involve review of site data, regional information, and case studies. In planning exercises for major restoration projects, experimental manipulations may also be conducted on a microcosm (e.g., laboratory) or mesocosm (e.g., field) scale to aid in understanding key parameters, processes, and potential pitfalls. Experimental studies may be especially important during feasibility studies.

DEFINING OBJECTIVES

Defining project objectives is the most important single step in the planning process because it ensures that a "road map" for the project is in place. To define the objectives, the site ecosystem and its historical development must be understood first. The depth of understanding necessary will vary by site, but the section on the relevant ecosystem type(s) in the *Ecosystem and Restoration Profiles* chapter of this report is a good guide to what topics should be evaluated. Key concepts of ecological theory that should be applied to each site are discussed in the next chapter. In most instances, site-specific baseline studies will be required along with the professional judgment of scientists with experience at the site or in similar habitats. Often, developing an understanding of the site is an

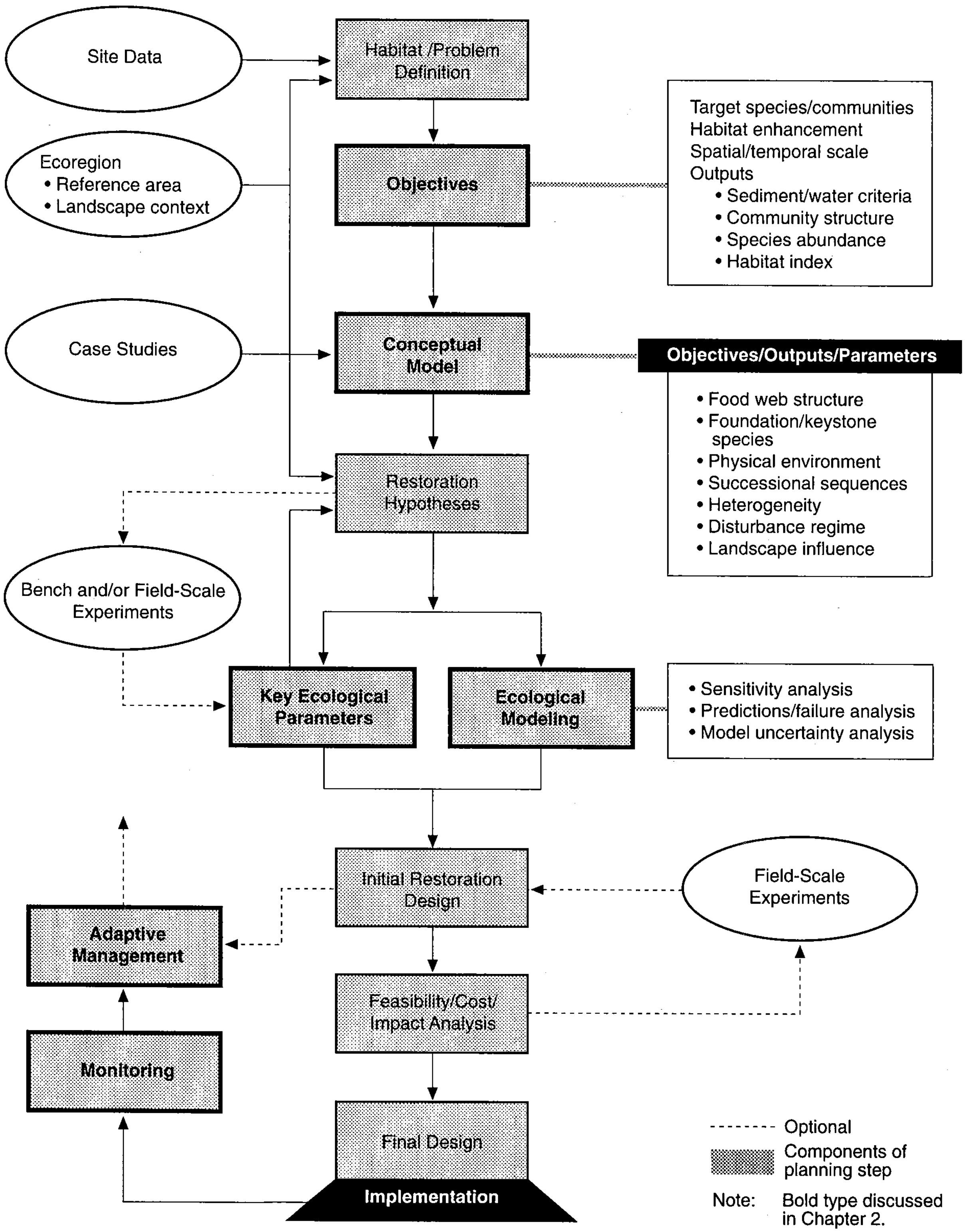

FIGURE 2-1. Ecological planning process for environmental restoration projects.

iterative process, increasing in scope and detail as one progresses through the restoration process.

Second, the site problem must be adequately understood, and it usually must be understood from both a scientific perspective and a broader "stakeholders" perspective. The scoping process used for environmental impact statements is a good analog to this problem definition process, although the specific stakeholders may differ. At this point, restoration objectives can be defined that specify:

- Target species, biological communities, or abiotic functions to be restored
- Site or habitat characteristics to be enhanced
- Spatial and temporal scale of restoration
- Outputs (i.e., performance indicators).

Objectives of the restoration project should be as specific as possible. Although an initial statement of a general narrative objective (or goal, such as "restoring water quality"; NRC 1992) may be useful, specific objectives are needed to maximize project success. NRC (1992) distinguishes goals from objectives, with the latter being more specific (often quantitative) statements of characteristics to be achieved in the restoration projects. For example, a goal for a lake restoration project could be "to enhance water quality" and the objectives related to this goal would specify the exact characteristics of water quality to be achieved, such as "increase water clarity by 20 percent." Specific objectives could then define a suite of habitat quality indicators beneficial to the desired biological community. This process is discussed in detail in the goal setting and adaptive management chapter.

In most cases, specific objectives and quantitative criteria for performance indicators can be defined without stating the more general goals that are implied by the objectives. These quantitative criteria are either expressions of relative change in an indicator (e.g., a "20 percent" increase in water clarity) or absolute numerical targets to be achieved (e.g., water and sediment quality criteria, species abundance and biomass levels, species richness values, community structure index values, habitat index values).

Restoration of physical-chemical habitat conditions and/or biological components of an ecosystem may be included as objectives of a restoration project. In most cases, objectives that specify changes in habitat conditions will be related to enhancement of fish and wildlife species abundance. Target species for restoration may include foundation species, such as kelp or seagrass, which provide the key structural component of habitat for a variety of other species. Rare, threatened, or endangered species may be addressed in some restoration projects, in which case the project will generally be a cooperative effort with the U.S. Fish and Wildlife Service. If an entire biological community is targeted for restoration, indicator species that reflect the status of the community would typically be specified in the objectives.

In some cases, habitat conditions may be enhanced primarily for human aesthetic or cultural reasons, with detrimental consequences for certain ecosystem components. For example, increased water clarity in a lake may be desirable from a human aesthetic perspective, but this habitat change may be related to a reduction in primary productivity and fish biomass. In addition, introduction of harvestable species, such as largemouth bass, bullfrogs, or crayfish, for recreational uses may cause declines in native fish, amphibian, or invertebrate populations or restructuring of ecological communities. Any restoration objective that is related primarily to human aesthetics or consumptive uses should be evaluated carefully to determine the risk of adverse ecological effects.

U.S. EPA (1990) provides specific guidance on selection of indicators of ecological health in their environmental monitoring and assessment program, which is equally applicable to restoration objectives. Criteria for selecting ecological indicators include the following:

- Indicators must be biological and incorporate elements of ecosystem structure and function
- Indicators must be socially relevant, clearly connected to environmental values, and responsive to individual or cumulative effects of stressors
- Indicators must be sensitive to various levels of stress, but not overestimate impacts resulting from natural variation
- Indicators should require limited sampling effort and be cost effective and have precedent in other, successful monitoring projects.

U.S. EPA (1990) notes that using community process and rate measurements (primary production, respiration, and nutrient cycling) in monitoring studies does not help identify ecological alterations in their earliest stages.

The level of detail of the objectives and their attainability (i.e., as indicated by the effort and costs required to meet the objectives) are determined by the size and complexity of the ecosystem as well as the nature of environmental degradation. Thus, the project scale in terms of area to be restored, the time period over which restoration efforts should be effective, and the extent of the biological community targeted for restoration should be defined while developing objectives. Factors to be considered in defining the project scale include the influence of the surrounding landscape, the size of the area of existing degradation or disturbance, the area required for adequate monitoring of results, and the available budget (NRC 1992).

In defining the objectives of a restoration project, available site data on the current status of habitats (e.g., measures of habitat quality) and fish and wildlife species should be reviewed in a regional context using reference areas (Figure 2-1). A regional perspective is important for defining reference areas and evaluating the status of an ecosystem and the influence of the natural physical setting (e.g., regional climate, geology, hydrology). An ecologically healthy reference area provides essential information for any restoration project. Reference area characterization (Table 2-1) is used to define the potentially attainable conditions for the habitat to be restored and as a point of reference for evaluating project success. The status of the ecosystem of concern and the related problems to be addressed by restoration efforts are defined relative to the regional reference conditions. The behavior of natural reference ecosystems based on available monitoring data is also used to develop restoration hypotheses (Figure 2-1). For example, hypotheses about the dynamics of a target species in restored systems may be based on predictions derived from available data on that species in regional reference systems.

Selection of reference sites and development of reference databases can be made more efficient with careful planning. First, ecological regions must be delineated based on geologic, geomorphic, and climatic factors. Water bodies within the same watershed or in nearby watersheds with the same geomorphic setting and climatic influences as the restoration site will provide the most appropriate reference sites. Reference areas should represent the desired habitat quality and range of habitat types and conditions. Anthropogenic influences on a potential reference site should be evaluated prior to its use as a reference site. Evaluation of potential influences of human activities will facilitate selection of appropriate ecological indicators. For example, what initially appears to be an undisturbed lake may be receiving large amounts of sediment from erosional processes occurring as a result of large-scale, accelerated slope failure. While the slope failure may be far upstream from the lake, changes in sediment accretion rates within the lake could change biological communities, making this site unreliable as an reference area.

The regional perspective is also important for evaluating the influence of the landscape on a project and the consequences of restoration within a landscape context. Lake and stream ecosystems are strongly influenced by their watersheds (e.g., nutrient and sediment inputs), and unique biological communities may develop as a result of landscape influences. The spatial distribution and dynamics of habitat patches within a landscape affect the distribution and abundances of most wildlife species, especially wide-ranging species (e.g., birds, large terrestrial

TABLE 2-1. EXAMPLES OF OUTPUTS FOR AQUATIC RESTORATION PROJECTS

Structural Characteristics

- Water quality
  - Dissolved oxygen
  - Dissolved salts
  - Dissolved toxics and other contaminants
  - Floating and suspended matter
  - pH
  - Odor
  - Opacity
  - Temperature profiles
- Soil condition
  - Soil chemistry
  - Erodibility
  - Permeability
  - Organic content
  - Soil stability
  - Physical composition, including particle sizes and microfauna
- Geologic condition
  - Surface rock
  - Subsurface rock
  - Other strata, including aquifers
- Hydrology
  - Quantity of discharge on annual, seasonal, and episodic basis
  - Timing of discharge
  - Hydraulic processes, including velocities, turbulence, shear stress, bank/stream storage, and exchange processes
  - Groundwater flow and exchange processes
  - Retention times
  - Particle size distribution and quantities of bed load and suspended sediment
  - Sediment flux (aggradational or degradational tendencies)
- Topography
  - Surface contours
  - Relief (elevations and gradients) and configuration of site surface features
  - Project size and location in the watershed, including position relative to similar or interdependent ecosystems
- Morphology
  - Lake
    - Shoreline circumference-to-area ratio
    - Mean depth
    - Mean-depth-to-maximum-depth ratio
  - River and streams
    - Channel patterns (braided, meandering, or straight)
    - Bank width-to-depth ratios
    - Meander geometry (amplitude, length, radius of curvature)
    - Cross-sectional depth profiles
    - Riffle-to-pool ratio (river and stream descriptions)
  - Wetlands
    - Inlets and outlets
    - Channels
    - Islands
    - Water retention time
    - Adjacent uplands-to-wetlands ratio
    - Fetch and exposure
    - Vegetation-water interspersion
- Flora and fauna
  - Density
  - Diversity
  - Growth rates
  - Longevity
  - Species integrity (presence of full complement of indigenous species found on the site prior to disturbance)
  - Productivity
  - Stability
  - Reproductive vigor
  - Size- to age-class distribution
  - Impacts on endangered species
  - Incidence of disease
  - Genetic defects
  - Genetic dilution (by nonnative germ plasm)
  - Elevated body burdens of toxic substances
  - Evidence of biotic stress
  - Carrying capacity, food web support, and nutrient availability

Functional Characteristics

- Surface water and groundwater storage, recharge, and supply
- Floodwater and sediment retention
- Transport of organisms, nutrients, and sediments
- Humidification of atmosphere (by transpiration and evaporation)
- Oxygen production
- Nutrient cycling
- Biomass production, food web support, and species maintenance
- Provision of shelter for ecosystem users (e.g., from sun, wind, rain, or noise)
- Detoxification of waste and purification of water
- Energy flow

Emergent Properties

- Resilience
- Persistence
- Verisimilitude

Source: NRC (1992)

mammals, and marine mammals) that depend on landscape-scale resources (Turner and Gardiner 1991; Dunning et al. 1992). Thus, the status of a restored ecosystem (i.e., patch within a landscape) may contribute to or benefit from landscape-level resources and influence the distribution and abundance of regional species.

ECOLOGICAL MODELS, HYPOTHESES, AND KEY PARAMETERS

The objectives of a restoration project and information from site-specific surveys or case studies provide the basis for developing a conceptual model of the ecosystem (Figure 2-1). The conceptual model shows the relationships among target species (or communities), outputs, and key ecological parameters. Such a model forms the basis for developing restoration hypotheses, which are postulates that describe causal mechanisms that lead to changes in target species (or communities). Such a model can be based initially on information provided in the relevant ecosystem profile(s) section of this report, with the addition of site-specific information. Restoration hypotheses state the expected changes in performance indicators in relation to key ecological parameters, including those parameters that are manipulated in the restoration effort (Table 2-2).

Key ecological parameters are the driving variables that determine community structure and function and, by definition, influence performance indicators. Some variables, such as the abundances of foundation and keystone[1] species, may be both outputs and key ecological parameters. Figure 2-1 shows relationships among the objectives of a restoration project, the outputs used as performance indicators, and the key ecological parameters.

To the extent practical, ecosystem restoration projects should address the causes of degradation, not just the symptoms. The use of models aids in understanding ecological processes and in identifying key parameters that drive observed changes in ecological systems. Conceptual models may be used to identify foundation species, keystone species, and engineer[2] species; identify key ecological parameters; and develop quantitative ecological models.

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[1] Keystone species have a disproportionately large influence on the relative species abundance of a particular ecosystem (e.g., a predator that removes a dominant competitor at a lower trophic level). See discussion of these species in the next chapter.

[2] Engineer species modify or create habitat in a significant way by virtue of their life activities (e.g., beavers creating impoundments or carp disturbing bottom sediments during feeding).

TABLE 2-2. EXAMPLE HYPOTHESES FOR AQUATIC RESTORATION PROJECTS

Project Type	Problem	Action	Example Hypothesis
Seagrass bed restoration	Seagrass bed was destroyed by boat traffic, and habitat for a valuable fish species was reduced.	1) Control boat traffic. 2) Transplant seagrass to enhance habitat for target fish species A.	Fish species A will colonize the restored habitat after the density of transplanted eelgrass is greater than 20 shoots/m^2. Transplanted eelgrass will reach 20 shoots/m^2 within 2 years. Within 6 years, fish species A will reach densities and age structure comparable to populations in reference area unaffected by boat traffic.
Lake restoration	Eutrophication of a lake due to fertilizer runoff from agricultural areas caused seasonal anoxia in the hypolimnion and elimination of salmonids.	1) Control nutrients in runoff from agricultural areas using nonpoint source strategy. 2) Install hypolimnetic aerator. 3) Stock juveniles of target salmonid species.	Target salmonid species will establish a self-reproducing fishery within 3 years.
Cattail marsh restoration	Elimination of beaver resulted in loss of cattail marsh habitat from a riparian wetland and invasion of alder and willow.	1) Phase I: Introduce beaver. 2) Phase II: Transplant cattails into portions of beaver ponds.	The extent of hydric soils and communities of hydrophilic plants (e.g., skunk cabbage, sedges, rushes, cattail) will cover >50 percent of the riparian wetland within 5 years. These plant communities will be maintained naturally as long as the beaver populations are undisturbed.
Stream restoration	Streambed is continually eroded because large woody debris was removed to facilitate boat traffic. Modification of the streambed led to loss of riparian wildlife habitat.	Introduce large woody debris at appropriate locations. Debris should consist of a mix of logs of varying diameter and some whole trees with DBH >0.5 m and intact root systems.	Woody debris will stabilize the streambed by reducing erosion of sediment. Debris will also enhance pool habitats and facilitate development of riparian marshes. Diversity of birds and mammals in riparian habitat will increase to within 70 percent reference site levels within 10 years.

Note: DBH - diameter at breast height

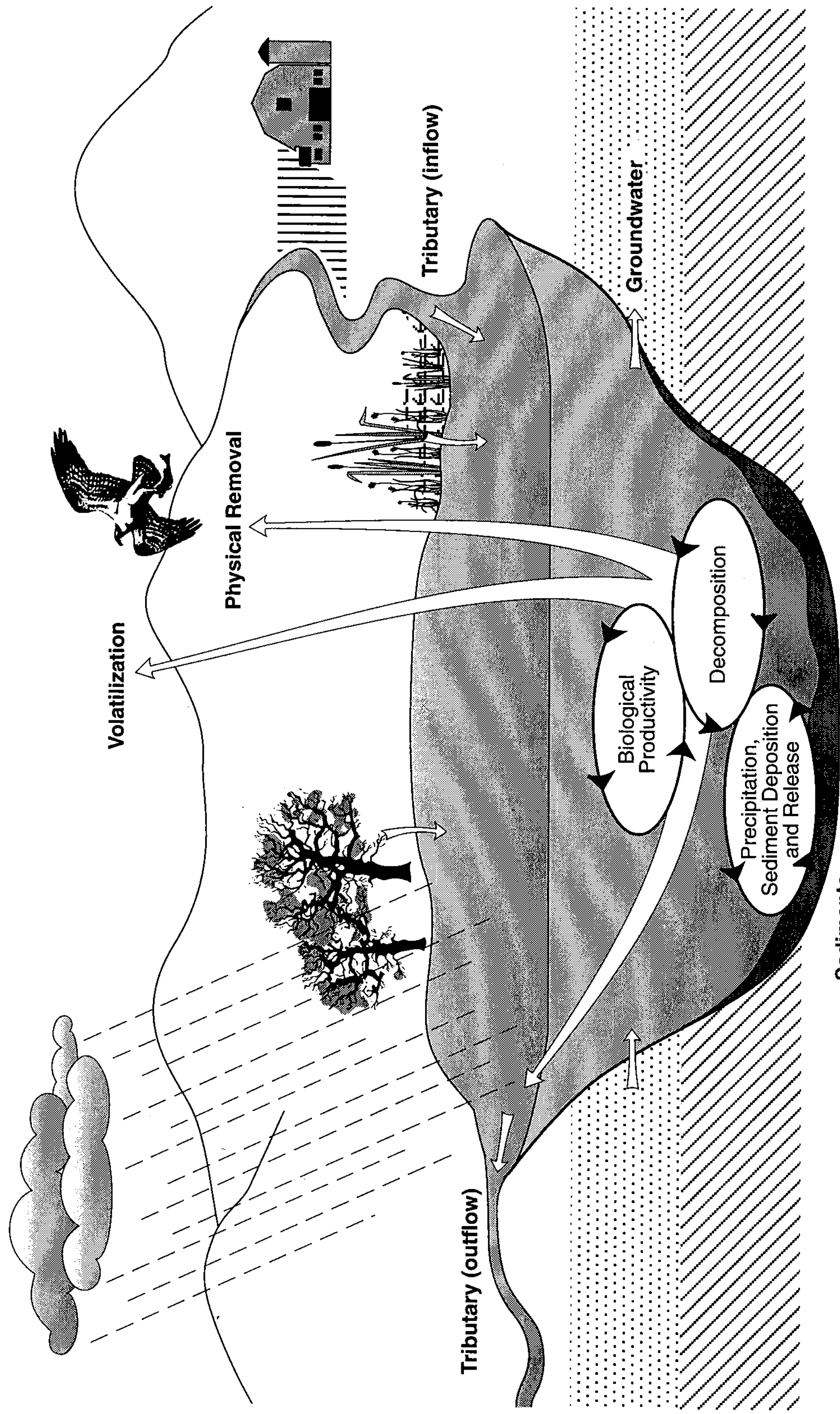
The components of a conceptual ecosystem model may include the following:

- Key abiotic processes or habitat characteristics
- Food web structure and key resource species
- Foundation, keystone, and engineer species that may affect the restoration goal (if known; see discussion of models below)
- Optimal physical characteristics to satisfy restoration goal
- Successional sequences after natural or anthropogenic disturbance
- Spatial and temporal heterogeneity in habitat that may affect restoration goal
- Natural disturbance regime that affects the restoration goal
- Landscape influences that support or inhibit the restoration goal.

If possible, the conceptual model should be summarized in a diagram or series of diagrams that illustrates basic relationships among ecosystem components, including key processes. Such diagrams may include flow charts that illustrate ecosystem compartments and processes (e.g., Figure 2-2), relatively detailed food webs (e.g., Figure 2-3), and fault-trees that show mechanisms of environmental degradation (e.g., Figure 2-4) or enhancement. Fault-trees may also be used to evaluate potential failures of restoration actions and to perform uncertainty analyses.

The restoration hypotheses and the identification of key ecological parameters leads to the design of restoration actions (Figure 2-1). Examples of key parameters for habitats of interest are shown in Table 2-3. Quantitative ecological models are valuable for refining hypotheses and ranking key ecological parameters. For example, model sensitivity analysis may aid in identifying parameters that most influence performance indicators. Quantitative uncertainty analysis and failure analysis are useful for evaluating alternative restoration actions and for avoiding "pitfalls."

SAIC (1996) describes available ecological models that are potentially useful in planning and evaluating restoration projects. In addition, bench-scale, plot-scale, or demonstration-scale experiments may be valuable for validating models and for performing empirical sensitivity analysis to identify key ecological parameters. Tanner et al. (1994) illustrate the use of a combination of empirical community-level studies, modeling, and sensitivity analysis to characterize ecological systems and identify keystone species.



Source: Modified from Herdendorf et al. (1992).

FIGURE 2-2. Example diagram of materials flux within lakes and reservoirs.

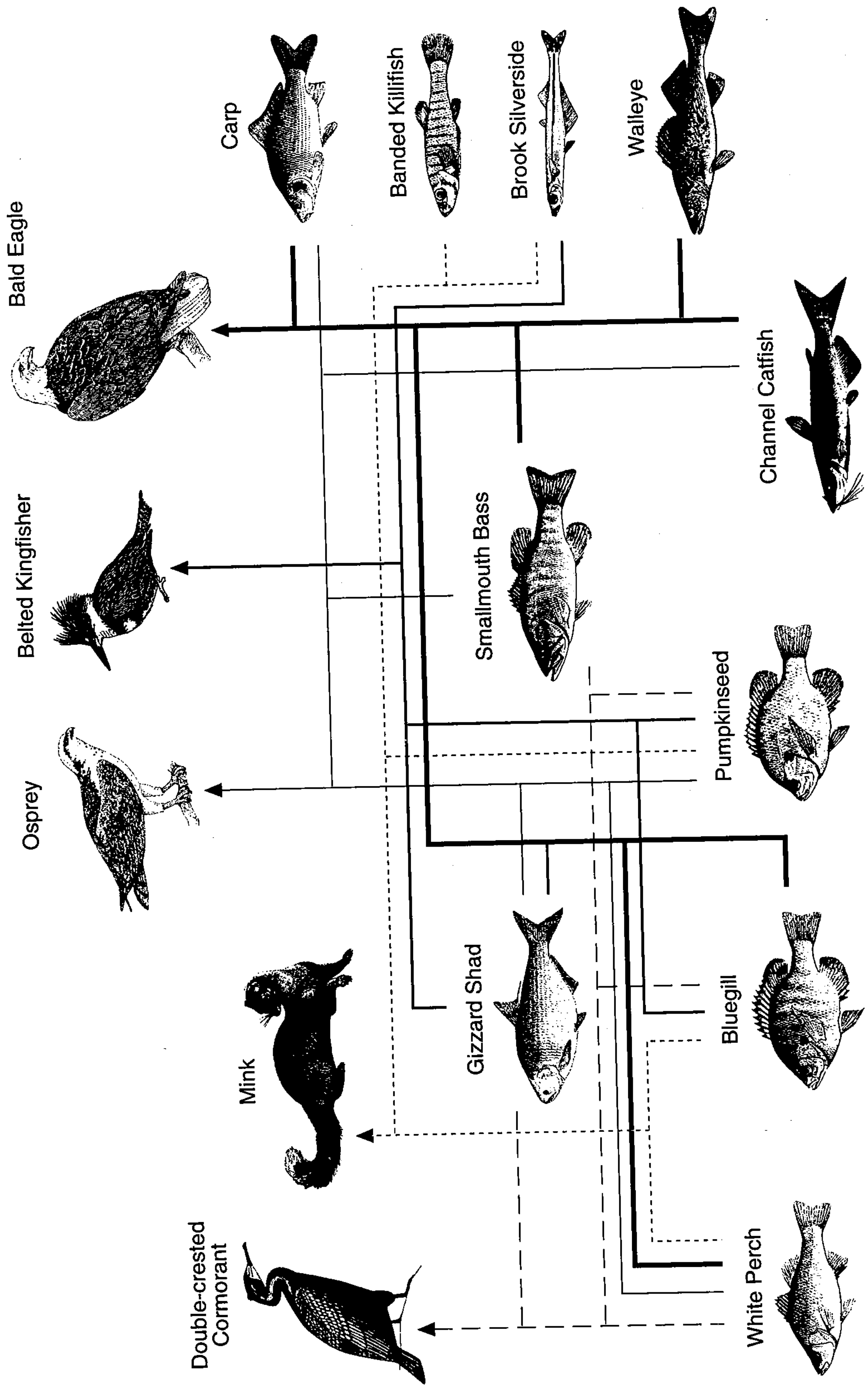
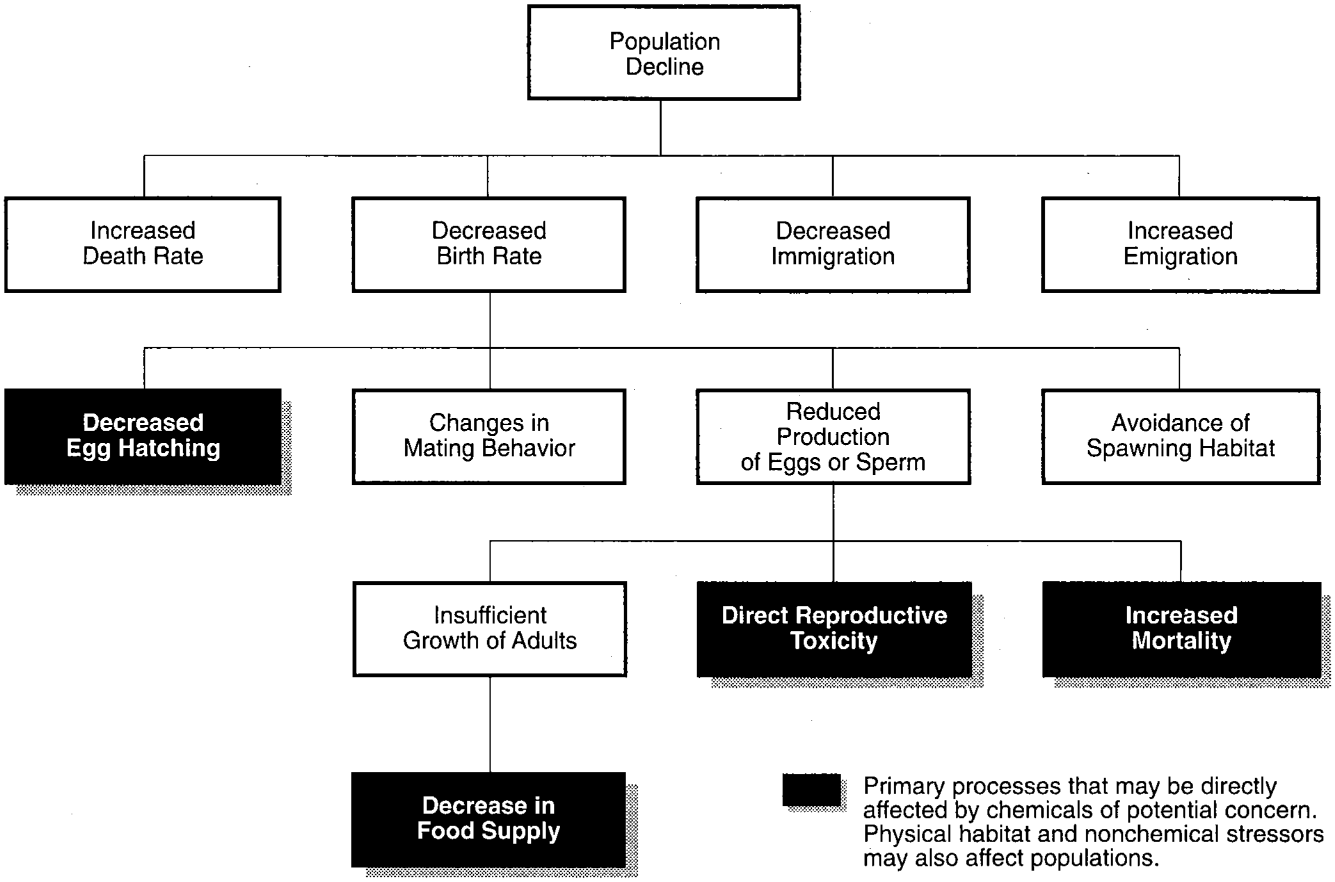


FIGURE 2-3. Example food web for lake in northeastern United States.



Source: Modified from Suter (1993).

FIGURE 2-4. Example fault tree analysis for population decline due to toxic chemicals.

TABLE 2-3. EXAMPLES OF KEY PARAMETERS FOR AQUATIC HABITATS

Habitat	Key Parameters
Open coastline and near coastal waters	■ Water quality ■ Tidal patterns and wave action ■ Substrate type
Subtidal estuarine habitats	■ Substrate type, grain-size distribution ■ Wave and current energy ■ Redox gradients in sediments ■ Nutrient cycling ■ Contaminant load ■ Solid-dissolved biogeochemical interactions ■ Provision of shelter/refuge and attachment substrate for benthic organisms ■ Availability of detritus ■ Natural disturbances due to storms, floods, ice scour, geologic events, sedimentation, and sea level change
Estuarine and marine wetlands	■ Water depth ■ Circulation/current velocity ■ Turbidity ■ Substrate quality ■ Salinity ■ Redox potential ■ Competition between plant species ■ Plant and animal interactions
Freshwater wetlands	■ Hydroperiod/hydrodynamics/water source ■ Flow velocity ■ Land surface elevation/soil moisture regime ■ Substrate type ■ Proportion of open water habitat ■ Nutrient availability ■ Dissolved oxygen within open-water habitat ■ pH of soil ■ Structural diversity ■ Disturbance type, frequency ■ Landscape position ■ Type, quality, and quantity of nearby wetlands
Streams and rivers	■ Flow ■ Stream channel morphology ■ Temperature ■ Light intensity ■ Toxic chemical concentrations ■ Species richness and abundance ■ Primary productivity ■ Riparian vegetation community type ■ Riparian corridor species composition ■ Flood frequency and intensity

TABLE 2-3. (cont.)

Habitat	Key Parameters
Lakes and reservoirs	■ Erosion of littoral substrate ■ Turbidity ■ Macrophyte abundance and diversity ■ Macroinvertebrate abundance and diversity ■ Fish community structure ■ Substrate type ■ Water quality

DEALING WITH UNCERTAINTY AND FAILURE

Planning for potential failure of a project is perhaps the best strategy for maximizing project success. Random variability in space and time is an inherent part of ecosystems. Moreover, uncertainty in our basic understanding of ecological parameters and processes increases the risk of failure. Characterizing variability and uncertainty during project planning can lead to a better understanding of potential failure and they increase the chances of project success. Quantitative uncertainty analysis of any ecological model used during restoration planning helps to define the limits of our understanding. The "safe-fail" approach of restoration design is exactly counter to the traditional "fail-safe" approach of standard engineering design.

Assuming the project design is properly implemented, failure of a restoration project is generally related to inadequate objectives or a poor functional design (NRC 1992). For example, restoration projects may fail for the following reasons:

- Poor definition of the initial problem or inadequate understanding of undesirable ecosystem characteristics
- Vague or overly ambitious project objectives
- Inadequate understanding of nutritional requirements and tolerance limits of target species
- Attempts to stabilize the physical-chemical environment (i.e., problem of overcontrol) when the target species requires environmental variation
- Insufficient colonizers or transplants of appropriate genetic stock
- Invasion of exotic species or other undesirable species that outcompete the target species
- Unpredictable species interaction
- Unpredictable natural or anthropogenic disturbances.

The two strategies recommended for avoiding pitfalls and recovering from restoration project setbacks are "bet-hedging" during restoration project design and adaptive management during implementation and monitoring (see Chapter 4 for discussion of this latter approach). The uncertainty inherent in forecasting future environmental conditions for a specific restoration project precludes selection of a single set of values for design parameter as the "optimal design." Thus, part of the planning of an optimal design should incorporate heterogeneity in the physical, chemical, and biological design factors as a bet-hedging strategy.

For example, the restoration area may be divided into spatial modules and several restoration subdesigns may be spatially distributed across the entire area. Holling (1978) and Walters (1986) provide details on adaptive environmental management. Suter et al. (1993) discuss methods for ecological risk assessment as applied to a variety of environmental problems.

Because future behavior of the system is never completely and accurately predicted by the planning process, spatial variation in design accomplishes two things. First, it increases the probability that at least one of the subdesigns is successful, creating a viable habitat "island." Propagules may then spread from this habitat island. Many engineer species modify their environment in such a way that colonization by additional individuals and expansion of the habitat is enhanced (e.g., in establishing a seagrass bed, the presence of seagrass dampens wave action, increases sediment deposition, and favors further colonization at the edge of the bed). Second, even if most of the subdesigns fail initially, this strategy results in information on which series of subdesigns are most effective. Such information is key to using an adaptive management approach. If most of the subdesigns succeed initially, future environmental variations may still lead to failure or disturbance of some of the subdesigns or of a portion of the restoration area. Spatial variation in the design may confer some resistance to unforeseen natural disturbances as well as resilience following disturbance.

RESTORATION DESIGNS, FEASIBILITY, AND EXPERIMENTATION

Restoration design may be developed from conceptual models, quantitative models, and data from previous case studies or restoration experiments. Restoration designs include engineering plans and constraints, which are being addressed in a separate work unit under the Evaluation of Environmental Investments Research Program (EEIRP). Confidence in the restoration design will generally be higher as more data are used to support the evaluation of alternatives in the feasibility study. The results of any experiments should be used in developing the initial design and the feasibility study (Figure 2-1). Experiments may also be conducted specifically to support the feasibility, cost, and impact analysis, especially tests of alternative designs at the larger scales.[3] Such experiments may also be used to refine (or validate) ecological models and restoration hypotheses (e.g., feedback loops in Figure 2-1).

Local and regional ecological constraints on restoration efforts must be considered in the evaluation of feasibility, cost, and impacts of each alternative. The

[3] Economic analysis is beyond the scope of the present discussion, but is addressed in a separate work unit of the EEIRP.

quantitative objectives of the project should be realistic relative to these constraints. Assuming that a design is properly implemented, restoration efforts may still have adverse effects on nontarget species. There is also some risk of project failure, with adverse consequences for the target species. Thus, risks to both target and nontarget species should be evaluated for each restoration alternative.

Multiattribute decision techniques may be useful for evaluating the benefits and risks of restoration alternatives (Brown et al. 1980; Edwards and Newman 1982; Brown and Valenti 1983; Tetra Tech 1986). Technical factors that should be considered in evaluating restoration alternatives or combinations of alternatives include:

- Expected benefits to target species and nontarget species within the project area
- Expected benefits to other systems in the local landscape and region
- Risk of adverse impacts on nontarget species (e.g., effects on threatened or endangered species)
- Past successes and failures with restoration design or similar efforts (i.e., case study analysis and experimental studies)
- Mechanisms for failures and their probabilities (i.e., fault-tree analysis)
- Relative sensitivity of ecosystem components to physical impacts of engineering efforts
- Initial ecosystem manipulation and degree of maintenance needed (i.e., ability of restored ecosystem to be self-sustaining)
- Projected time until specific target conditions are attained
- Influence of other local projects (e.g., housing development, road construction) on habitat recovery time or potential outcomes.

Optimal physical-chemical conditions and genetically optimal stocks of target species should be used in restoration projects. Optimality is defined primarily in terms of the stated objectives and outputs of the project during the planning process but may include similarity with reference site conditions of physiochemical condition, or similarity to historic genotypes. The conditions and values of key parameters defined as important for achieving objectives are based on past experience. Nevertheless, a timely optimal design may include several subdesigns with minor variations that maximize the potential for success in the face of environmental unpredictability.

Mesocosm-scale (e.g., plot- or enclosure-scale) tests of restoration designs or subdesigns may provide experimental verification that a design will work, at least for some portion of the restoration area. Mesocosm testing must be done at an appropriate spatial scale. The choice of spatial scale for mesocosms depends on a balance between making the mesocosm large enough to provide realistic results that are not confounded by scale effects and small enough to be feasible and cost-effective. Voshell (1989) and Graney et al. (1994) discuss past experiences with mesocosms in ecological research and recommendations for mesocosm design.

IMPLEMENTATION, MONITORING, AND ADAPTIVE MANAGEMENT

A final ecological restoration design is developed from the results of the feasibility, cost, and impact analysis (Figure 2-1). If mesocosm testing is done, the results may be used to refine ecological models before the final restoration design is developed. Implementation of the design is covered by the environmental engineering work unit within EEIRP.

The physical, chemical, and biological characteristics of the manipulated ecosystem should be monitored to measure the success of the restoration effort and to perform adaptive management actions, if necessary. Adaptive management (e.g., as described in Holling [1978]) recognizes that management techniques can and should evolve as new information is received. The key purpose of monitoring with respect to adaptive management is twofold. First, monitoring guides further selective manipulations of the project that improve the outcome relative to stated objectives. Second, monitoring allows evaluation of the effectiveness of specific restoration methods or techniques. These can then be used with greater certainty on other projects. Design of monitoring programs is a complex topic that has been addressed by many authors (e.g., U.S. EPA 1990; Green 1979; Mar et al. 1986; Loeb and Spacie 1994). The discussion here focuses on selection of the ecosystem characteristics to be monitored (i.e., outputs; also termed assessment criteria by NRC 1992).

One of the most difficult decisions in designing a monitoring program for a restoration project is the initial choice of outputs (i.e., the structural or functional elements of the ecosystem that are to be used to judge the success of the project) (NRC 1992). These outputs should be linked to project objectives within the context of known attributes of regional reference ecosystems. Multiple outputs should be used to minimize the risk of missing important ecological effects of the project (NRC 1992). Examples of general assessment criteria are summarized in Table 2-1.

Restoration project planning actually starts with the definition of existing problems, a clear statement of project objectives, and an understanding of uncertainty. Recognition of these fundamental concepts should continue throughout the planning, design, implementation, and monitoring phases of the project. Table 2-4 shows a checklist that can be used to evaluate planning and management actions throughout all phases of restoration projects (NRC 1992). Additional discussion of goal setting and adaptive management is provided in Chapter 4.

TABLE 2-4. RESTORATION PROJECT PLANNING AND IMPLEMENTATION CHECKLIST

Project Planning and Design

1. Has the problem requiring treatment been clearly understood and defined?
2. Is there a consensus on the mission of the restoration program?
3. Have the goals and objectives been identified?
4. Has the restoration been planned with adequate scope and appropriate expertise?
5. Does the restoration management plan have an annual or midcourse correction point in line with adaptive management procedures?
6. Are the outputs—the measurable biological, physical, and chemical attributes—directly linked to the objectives?
7. Have adequate monitoring, surveillance, management, and maintenance programs been developed along with the project, so that monitoring costs and operational details are anticipated and monitoring results will be available to serve as input in improving restoration techniques used as the project matures?
8. Has an appropriate reference system(s) been selected from which to extract target values of performance indicators for comparison in conducting the project evaluation?
9. Have sufficient baseline data been collected over a suitable period of time on the project ecosystem to facilitate before-and-after comparisons?
10. Have critical project procedures been tested on a small experimental scale in part of the project area to minimize the risk of failure?
11. Has the project been designed to make the restored ecosystem as self-sustaining as possible to minimize maintenance requirements?
12. Has thought been given to how long monitoring will have to be continued before the project can be declared effective?
13. Have risk and uncertainty been adequately considered in project planning?

During Restoration

1. Based on the monitoring results, are the anticipated intermediate objectives being achieved? If not, are appropriate steps being taken to correct the problem(s)?
2. Do the objectives or performance indicators need to be modified? If so, what changes may be required in the monitoring program?
3. Is the monitoring program adequate?

Post-Restoration

1. To what extent were project goals and objectives achieved?
2. How similar in structure and function is the restored ecosystem to the reference ecosystem?
3. To what extent is the restored ecosystem self-sustaining, and what are the maintenance requirements?
4. Have critical ecosystem functions been restored?
5. Have critical components been restored?
6. How long did the project take?
7. What lessons have been learned from this effort?
8. Have those lessons been shared with interested parties to maximize the potential for technology transfer?
9. What was the final cost, in net present value terms, of the restoration project?
10. What were the ecological, economic, and social benefits realized by the project?
11. How cost-effective was the project?
12. Would another approach to restoration have produced desirable results at lower cost?

Source: NRC (1992).

REFERENCES

Brown, C.A., and T. Valenti. 1983. Multi-attribute tradeoff system: user's and programmer's manual. U.S. Department of the Interior, Bureau of Reclamation, Denver, CO.

Brown, C.A., R.J. Quinn, and K.R. Hammond. 1980. Scaling impacts of alternative plans. U.S. Department of the Interior, Bureau of Reclamation, Denver, CO.

Dunning, J.B., B.J. Danielson, and H.R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. Oikos 65:169–175.

Edwards, W., and J.R. Newman. 1982. Multiattribute evaluation. Sage Publications, Beverly Hills, CA.

Graney, R.L., J.H. Kennedy, and J.H. Rodgers. 1994. Aquatic mesocosm studies in ecological risk assessment. Special publication of the Society of Environmental Toxicology and Chemistry.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley & Sons, New York, NY.

Herfendorf, C.E., L. Håkanson, D.J. Jude, and P.G. Sly. 1992. A review of the physical and chemical components of the Great Lakes: a basis for classification and inventory of aquatic habitats. pp. 109–160. In: The Development of an Aquatic Habitat Classification System for Lakes. W.D.N. Busch and P.G. Sly (eds). CRC Press, Boca Raton, FL.

Holling, C.S. 1978. Adaptive environmental assessment and management. John Wiley & Sons, London, England.

Loeb, S.L., and A. Spacie. 1994. Biological monitoring of aquatic systems. Lewis Publishers, Boca Raton, FL.

Mar, B.W., R.R. Horner, J.S. Richey, R.N. Palmer, and D.P. Lettenmaier. 1986. Data acquisition. Environ. Sci. Technol. 20:545–551.

NRC. 1992. Restoring aquatic ecosystems: science, technology, and public policy. National Research Council Commission on Geosciences, Environment, and Resources. National Academy Press, Washington, DC.

SAIC. 1996. Restoration analysis: use of models to predict restoration success. Prepared for U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS. Science Applications International Corporation, Bothell, WA.

Suter, G.S., II (ed). 1993. Ecological risk assessment. Lewis Publishers, Boca Raton, FL.

Tanner, J.E., T.P. Hughes, and J.H. Connell. 1994. Species coexistence, keystone species, and succession: a sensitivity analysis. Ecology 75(8): 2204–2219.

Tetra Tech. 1986. A framework for comparative risk analysis of dredged material disposal options. Prepared for Puget Sound Dredged Disposal Analysis, U.S. Army Corps of Engineers, Seattle District. Tetra Tech, Inc., Bellevue, WA.

Turner, M.G., and R.H. Gardiner (eds). 1991. Quantitative methods in landscape ecology: the analysis and interpretation of landscape heterogeneity. Springer-Verlag, New York, NY.

U.S. EPA. 1990. Environmental monitoring and assessment program, ecological indicators. EPA/600/3-90/060. C.T. Hunsaker and D.E. Carpenter (eds). U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Voshell, Jr., J.R. (ed). 1989. Using mesocosms to assess the aquatic ecological risk of pesticides: theory and practice. Miscellaneous Publication #75. Entomological Society of America.

Walters, C.J. 1986. Adaptive management of renewable resources. McGraw-Hill, New York, NY.

3. INCORPORATING ECOLOGICAL THEORY INTO RESTORATION PROJECT PLANNING

Jennifer Sampson, Anne MacDonald, and Robert Pastorok

Ecological theory relates biotic and abiotic processes occurring on different scales of space, time, and organizational complexity and provides an understanding of patterns in terms of the processes that produce them (Levin 1992). This conceptual framework is essential for effective resource management and habitat restoration planning. This chapter focuses on ecological concepts that are particularly important to consider in restoration planning without attempting to review ecological theory in total (Odum [1971] and Krebs [1985] are useful introductions to ecological theory). Ecosystem components can be divided into two types: structural and functional. Structural components are the abiotic materials, biota, and their pattern within the ecosystem, including their spatial arrangement and food web relationships. Functional components are the interactions among the structural components (e.g., energy, material, and species fluxes). Because restoration by necessity can rebuild only the structure of ecosystems (in a way that optimizes ecosystem function) rather than rebuilding ecosystem function directly, the concepts discussed below are primarily related to the development of ecosystem structure over time and space. With these concepts, one can better understand the responses of biological communities to the physical manipulations of ecosystem structure inherent in any restoration activity.

PHYSICAL HABITAT VARIABILITY

If control of ecological processes is viewed as a hierarchy of environmental factors, large-scale physical parameters, such as climate and geology, would impart the greatest control on systems, followed by geomorphology and hydrologic patterns (e.g., Kellerhals and Church 1989; Naiman et al. 1992). Biological community structure and diversity is controlled on a smaller scale by immediate physical characteristics such as substrate quality, local hydrology, and the physical habitat provided by plant communities. Niche diversification (i.e., the availability of different types of ecological roles for species) within communities is directly related to variation in habitat structure (e.g., temperature, water chemistry, foundation macrophyte assemblage) and influences the number of species able to inhabit an area (Huston 1994).

All environments exhibit both spatial and temporal variation in physical parameters (DeAngelis 1994; Karr et al. 1992). Alterations in the suite of key physical parameters within an ecosystem will result in a shift in biotic communities, possibly causing feedbacks leading to further changes in community structure. Temperature, nutrient flux, hydrologic variables (e.g., precipitation, flooding, water level fluctuations), light availability, and water chemistry are physical-chemical parameters showing considerable variation in time and space. These and other parameters exert controls on resource availability, diversity of refugia (i.e., suitable habitats within an otherwise altered landscape), and energy and materials transport. In response to physical variation, patterns of primary productivity, the diversity and abundance of invertebrates, the structure of fish communities, and habitat utilization vary, initiating patterns of biological feedback. Populations respond in non-linear ways to environmental variation (Huntly 1995). The result is a constantly shifting mosaic of habitats, communities, and ecological processes in any given environment. In this context, identification of deterministic relationships and target process rates can be complex. Nonetheless, recognition and understanding of natural variation in each individual aquatic system allows a more informed selection of target variables for restoration.

LANDSCAPE ECOLOGY

The term landscape refers to a mosaic of habitat patches in which a particular patch (the focal patch) is located (Dunning et al. 1992). Landscape size is species-specific and is usually intermediate between an organism's home range and its regional distribution.

Landscape ecology employs a "big-picture" view in seeking to understand ecosystem functions as they relate to a heterogeneous mosaic of patches of habitat (i.e., ecosystem structure). Such a landscape perspective has been implicit in much of ecological research over the last 5 decades but is more widely applied now because of a convergence in major issues in land management and tools by which to analyze these issues. For instance, conservation of regionally important endangered species in this country (e.g., the California condor, manatee, grey wolf, Pacific salmon) and abroad (e.g., large African mammals) has required an understanding of how these animals use large areas of land, composed of many different natural and severely altered ecosystems. Second, improved computing has allowed for the numerical representation and manipulation of spatial data using, for instance, digital satellite "images" and geographic information systems.

The following elements are used to develop an understanding of landscape ecology:

- The development and dynamics of spatial heterogeneity
- Species and community interactions within heterogeneous landscapes

- The influence of spatial heterogeneity on biotic and abiotic processes
- The management of spatially heterogeneous environments (Turner and Gardiner 1991).

The distribution of resources between landscape features in aquatic habitats plays a role in structuring population dynamics and communities; for example, littoral vegetation in lakes attracts spawning fish, numerous invertebrate species, and semi-aquatic mammals (e.g., muskrat). Variation in landscape composition affects the distribution of resources in space such that patches provide several distinct services for one or more species or life stages. The availability or spatial distribution (accessibility) of each required patch type will affect population viability. The development of a clear understanding of the interactions within and between the patches provides the key to understanding landscape ecology.

When restoring habitat for a particular population, modeling resource distribution at the landscape scale may help prevent oversight of critical habitat requirements. At regional scales, aquatic habitats may be viewed as patches and/or corridors, affecting the ability of species to recolonize restored systems. For example, lake systems are often linked by surface water streams, which act as corridors for dispersal of some fish species (including exotic species). However, if passage between lakes is blocked by the presence of predators, disease, or physical habitat alteration, the lake with fish cannot be counted on as a biological source. Moreover, undesirable species in one lake may have to be controlled if a dispersal corridor exists linking that lake to a lake targeted for restoration. Knowledge of landscape ecology facilitates recognition of linkages between habitats and the distribution of resources. The use of the techniques described above to develop a better understanding of landscape ecology should lead to a better allocation of project funds.

Watershed Perspective

The watershed provides a useful landscape unit within which processes relevant to management of aquatic systems can be viewed. Aquatic ecosystems are shaped by processes occurring within their watersheds and reflect the cumulative impacts of hydrologic and geologic events occurring upslope (Naiman et al. 1992). Less commonly, processes occurring downstream in a watershed (e.g., channel instability) can propagate upstream along river courses. Effective management of aquatic ecosystems must always include evaluation of current and past development, land use, agricultural and industrial activities, and other large-scale human activities within the watershed, as well as an understanding of the basic hydrology and geomorphology.

Aquatic-Terrestrial Ecotones

Transitional zones between adjacent biological communities, called ecotones, represent very important parts of the ecological landscape. An ecotone may contain biotic and abiotic components found in either or both of the adjacent communities (Risser 1990). Ecotones are dynamic areas on the landscape with unique properties that are defined by the strength of interaction with adjacent habitats. Ecotones may control or facilitate the movement of organisms, materials, and energy between adjacent communities, may be foci of productivity on the landscape, and may contain unusually high species richness (Naiman and Décamps 1990). Definition of aquatic-terrestrial ecotones is scale specific, but examples include wetland margins, riparian forests, and floodplains. Recognition of the role of ecotones as functional zones of transition between communities may allow for more efficient use of ecotones in habitat restoration.

SPECIES INTERACTIONS

Species interactions are fundamental to the development of dynamic equilibria in ecosystems. When species interactions are disturbed, the structure of the community may shift or a cascade of effects may be observed. When planning restoration projects, key species interactions should be evaluated to identify important structural elements that should be carefully managed during manipulation of the abiotic or biotic components of the system. Some of the more important species interactions are described for biological communities by habitat in the ecological profile sections of this report. These interactions can be grouped into four patterns: competition, predation, coevolution, and symbiosis.

Competition

Competition occurs when two populations use the same resources and those resources are limited in the environment or through direct interference of one population by another (Pianka 1978). The result is a reduction in fitness of individuals in one or both populations. Competition may occur between individuals of the same species or between different species. In the long term, competition results in niche separation, diversification, and specialization (Pianka 1978). In the short term, steps in restoration projects may be necessary to moderate the competitive advantage of more aggressive species.

Predation

Predation is when an individual predator kills an individual prey for the purpose of consuming it. Like competition, predation is a key factor driving development of community structure (Paine 1969) and the evolution of species (May 1976).

Coevolution

Coevolution is the joint evolution of taxa such that they are closely related but do not exchange genes (Pianka 1978). Examples include plants and herbivores (e.g., herbivores as seed dispersers, the production of chemical defenses against herbivory by plants) or plants and pollinators. Restoration project planning should provide for specific coevolved species where necessary.

Symbiosis

Symbiotic relationships refer to pairs of organisms living together without harm to either player (therefore does not include parasitism and amensalism). Symbiosis includes mutualism (obligatory and non-obligatory), where both species benefit from the relationship, and commensalism, where one species benefits and the other is not affected by the relationship.

KEYSTONE SPECIES AND ECOLOGICAL ENGINEERS

A keystone species is one that has major effects on species composition in its community, usually through trophic interactions (Paine 1969). For example, removal of sea otters (the keystone species) from kelp forests allows sea urchin populations to expand. Because sea urchins consume kelp, kelp growth becomes severely attenuated, which in turn increases the effects of wave action resulting in greater disturbance of the sea bed (Lawton and Jones 1995). Because the presence of a keystone species shapes a community, its removal or reduction results in shifts in species abundances and community composition.

The concept of ecosystem engineer species is similar, but engineer species cause physical changes in their environment, which alter local habitats. Organisms that alter the environment through their own physical structure are called autogenic engineers (e.g., kelp, trees, littoral macrophytes), while those that transform energy or materials flux through mechanical or other means are allogenic engineers (e.g., beavers, sea urchins) (Lawton and Jones 1995). The scale of impact of the engineer depends on its population density, the spatial distribution of the population, the length of time the population has been present, the durability of constructs, the number and type of resource flows that are modified, and the number of species connected to modified resources.

There are few documented examples of keystone species, possibly because they are unrecognized, but examples of keystone species are believed to be common in aquatic communities (Krebs 1985; Grimm 1995). Understanding the impact and potential uses of keystone species may simplify restoration or enhance its potential for success. Ecosystem engineer species are found in almost every type of habitat (Lawton and Jones 1995). Restoration planning must evaluate the degree to which these species can modify site conditions and address probable impacts of these alterations. Where possible, these species should be used to further the site restoration process.

ROLE OF DISTURBANCE

Ecologists are beginning to recognize "disturbance" as an important force shaping ecological systems, but the term is applied loosely and is often poorly defined (Poff 1992). Disturbances are perceived to be discrete events that result in killing, damaging, or displacing organisms, thereby creating opportunities for the establishment of new organisms (Sousa 1984) or that disrupt ecosystem, population, or community structure by altering resources, substrate, or the physical environment (White and Pickett 1985).

Disturbance events (e.g., storms, floods, fire, tree blow-down) play a key role in structuring ecological communities and landscape patches. Disturbance events create new habitats, change the degree of habitat patchiness, weaken the status of dominant species, or change the distribution of resources such that the relative abundances of species and their interactions shift radically in a very short time period. Many types of communities are shaped by disturbance, including riparian zones, estuarine and coastal wetlands, and major upland plant communities (e.g., tallgrass prairie, chaparral, lodgepole pine forests). Anthropogenic impacts often alter the frequency and intensity of disturbance to beyond the point at which native species can regenerate. Awareness of natural disturbance regimes is critical to effective resource management: a goal of restoration should be to restore the natural frequency and intensity of specific disturbance patterns.

Disturbance "regime" is a function of the statistical distribution of the frequency and intensity of individual disturbance events. Resh et al. (1988) add that, to constitute a disturbance, an event must be outside of the predictable range of occurrence because organisms would be adapted to consistent temporal patterns. Poff (1992), rejecting Resh et al.'s "predictability clause," emphasizes that a physical event such as an annual snowmelt flood may be a disturbance at one scale (e.g., stream sediment) but not at another (e.g, drainage basin). Thus, although the concept of disturbance is clear, the scale at which a specific disturbance occurs must be defined to avoid ambiguity.

Legacies

An ecological legacy is the physical or biological response to environmental variation or disturbance that remains after the disturbance or alteration has ceased. For example, the legacy of prolonged nutrient enrichment of a lake may include a surplus of organic detritus in sediments, lack of fish in the hypolimnion, or a characteristic bathymetry of littoral substrate related to high sedimentation rates during eutrophication. A legacy of land drainage in a wetland setting can include a soil seed bank of invasive plants that must be dealt with during site restoration. Ecological legacies may be more or less manageable. For example, nutrient removal technology is being developed to address lake eutrophication, but the introduction of aggressive exotic species often has far reaching legacies that are difficult to predict or impossible to reverse even after removal of the exotic species. Recognition of potential legacies from restoration efforts or of existing legacies from past impacts will guide selection of monitoring variables and may help predict secondary and tertiary effects of restoration actions.

Lag Times

Lag time is the period of time between a physical or biological alteration (e.g., disturbance) and measurable environmental response. The magnitude of the lag time depends on the selected response variable. Any individual perturbation or disturbance may give rise to a range of ecological effects, all with different lag times. Moreover, the response in two separate watersheds to the same type of perturbation may occur at different rates depending on key physical parameters such as flood regime (i.e., hydrograph), lake turnover time, tidal mixing rates, or geomorphic setting. Understanding lag time will improve predictions of the nature and magnitude of environmental response.

CONCLUSION: SPATIAL AND TEMPORAL SCALES

The common thread of this discussion has been the need to formally recognize differences in the spatial and temporal scales of different ecological processes. The observer of an ecological process or event brings to bear a perceptual bias on its interpretation. If the observer is an organism in the environment, its bias will be a function of its physical size, life history, and longevity. Patterns of response to environmental variation reflect processes (described above) occurring at different spatial and temporal scales (Levin 1992). Effective analysis of ecological processes at one scale may require understanding processes occurring at another or multiple other scales. Explicit recognition of both the spatial and temporal scales of ecological analysis is critical to prevent ambiguity in approaches, data needs, and restoration targets (Levin 1992).

REFERENCES

DeAngelis, D.L. 1994. Synthesis: spatial and temporal characteristics of the environment. pp. 307–320. In: Everglades, the Ecosystem and Its Restoration. S.M. Davis and J.C. Ogden (eds). St. Lucie Press.

Dunning, J.B., B.J. Danielson, and H.R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. Oikos 65:169–175.

Grimm, N.B. 1995. Why link species and ecosystems? A perspective from ecosystem ecology. pp. 5–15. In: Linking Species and Ecosystems. C.G. Jones and J.H. Lawton (eds). Chapman & Hall, New York, NY.

Huntly, N. 1995. How important are consumer species to ecosystem functioning? pp. 72–83. In: Linking Species and Ecosystems. C.G. Jones and J.H. Lawton (eds). Chapman & Hall, New York, NY.

Huston, M.A. 1994. Biological diversity: the coexistence of species on changing landscapes. Cambridge University Press, Cambridge, MA.

Karr, J.R., M. Dionne, and I.J. Schlosser. 1992. Bottom-up versus top-down regulation of vertebrate populations: lessons from birds and fish. pp. 243–286. In: Effects of Resource Distribution on Animal-Plant Interactions. Academic Press, Inc., New York, NY.

Kellerhals, R., and M. Church. 1989. The morphology of large rivers: characterization and management. pp. 31–48. In: Proc. of the International Large River Symposium. D.P. Dodge (ed). Can. Spec. Publ. Fish. Aquat. Sci. 106.

Krebs, C.J. 1985. Ecology: the experimental analysis of distribution and abundance. Third Edition. Harper & Row, New York, NY.

Lawton, J.H., and C.G. Jones. 1995. Linking species and ecosystems: organisms as ecosystem engineers. pp. 141–150. In: Linking Species and Ecosystems. C.G. Jones and J.H. Lawton (eds). Chapman & Hall, New York, NY.

Levin, S.A. 1992. The problem of pattern and scale in ecology. Ecology 73(6):1943–1967.

May, R.M. (ed). 1976. Theoretical ecology: principles and applications. W.B. Saunders Company, Philadelphia, PA.

Naiman, R.J., and H. Décamps (eds). 1990. The ecology and management of aquatic-terrestrial ecotones. United Nations Education, Scientific and Cultural Organization, Paris, France.

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Connor, P.L. Olson, and E.A. Steel. 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest coastal ecoregion. Chapter 6. In: Watershed Management. R.J. Naiman (ed). Springer-Verlag, New York, NY.

Odum, E.P. 1971. Fundamentals of ecology. Third Edition. W.B. Saunders, Philadelphia, PA.

Paine, R.T. 1969. The *Pisaster-Tegula* interaction: prey patches, predator food preference, and intertidal community structure. Ecology 50:950–961.

Pianka, E.R. 1978. Evolutionary ecology. Second Edition. Harper & Row Publishers, New York, NY.

Poff, N.L. 1992. Why disturbances can be predictable: a perspective on the definition of disturbance in streams. J. N. Am. Benthol. Soc. 11(1):86–92.

Resh, V.H., A.V. Brown, A.P. Covich, M.E. Gurtz, H.W. Li, G.W. Minshall, S.R. Reice, A.L. Sheldon, J.B. Wallace, and R. Wissmar. 1988. The role of disturbance in stream ecology. J. N. Am. Benthol. Soc. 7:433–455.

Risser, P.G. 1990. The ecological importance of land-water ecotones. pp. 7–22. In: The Ecology and Management of Aquatic-Terrestrial Ecotones. R.J. Naiman and H. Décamps (eds). United Nations Education, Scientific and Cultural Organization, Paris, France.

Sousa, W.P. 1984. The role of disturbance in natural communities. Ann. Rev. Ecol. System. 15:353–391.

Turner, M.G., and R.H. Gardiner (eds). 1991. Quantitative methods in landscape ecology: the analysis and interpretation of landscape heterogeneity. Springer-Verlag, New York, NY.

White, P.S., and S.T.A. Pickett. 1985. Natural disturbance and patch dynamics: an introduction. pp. 3–9 In: The Ecology of Natural Disturbance and Patch Dynamics. S.T.A. Pickett and P.S. White (eds). Academic Press, New York, NY.

4. GOAL SETTING AND ADAPTIVE MANAGEMENT

Ronald Thom

When marine monitoring programs were summarized by NRC (1990), they concluded that among the most important components of a sound monitoring program were 1) clear goals and objectives and 2) flexible programs that allow modifications of the project to adapt to changes in conditions or new information. These two components, goal formulation and development of an adaptive management plan, are key components of a restoration project plan for an aquatic system and should be developed very early in the planning phase. The process of defining goals and developing an adaptive management plan are summarized in this chapter.

RESTORATION GOALS

Goal formulation involves developing a vision and formal goal statement. The formal goals lead to the next part of the planning process, which is developing performance criteria or objectives. A restoration project begins initially with a proven or perceived need to repair damages suffered by an ecosystem. The perceived outcome of the restoration project is generally developed into a visual image (i.e., vision) for the project (Thom and Wellman, in press). The vision includes major features of the system, such as vegetation type and distribution, hydrology, and fisheries and wildlife resource use. The vision is generally refined through interaction among scientists, engineers, and the public and forms the basis of more clearly defined goals for the restoration project. In defining goals for the restoration project, it is important to make goals as simple and unambiguous as possible, relate goals directly to the vision for the project, and set goals that can be measured or assessed in a monitoring program (Thom and Wellman, in press).

An example of the process of setting goals comes from the Central and Southern Florida Project (Corps 1994). This federal project, first authorized in 1948 to provide flood control, water control and water supply, has ultimately resulted in degradation of the south Florida ecosystem. The need for restoration of the ecosystem was formalized into a goal statement in 1992, which recommended modifying the federal project for "...improving the quality of the environment, improving protection of the aquifer, and improving the integrity, capability, and conservation of urban water supplies affected by the project or its operation"

(Water Resources Development Act of 1992, P.L. 102–590). This goal statement contains many undefined and difficult to define terms (e.g., improving integrity, capability).

Through a scientific work group and with public involvement, the goals were refined into six planning objectives concerning the ecosystem, water quality, water supply, flood control, recreation, the economy and social considerations (Corps 1994). These planning objectives are as follows:

1. Increase the total spatial extent of wetlands
2. Increase habitat heterogeneity
3. Restore hydrologic structure and function
4. Restore water quality conditions
5. Improve the availability of water
6. Reduce flood damages on Seminole and Miccosukee tribal lands.

Along with providing guidance for designing the project, these goals form hypotheses against which performance of the restored system can be assessed. However, the goals lack the specificity of formal performance criteria. For example, "increase the spatial extent of wetlands" does not specify by how much. Furthermore, by literally interpreting terms that are presently in vogue, such as "habitat heterogeneity," the result may actually be a more fragmented and degraded landscape if carried to an extreme. Specificity and detail are developed as part of the performance criteria.

Guidance for developing performance criteria from goals has been developed under another Evaluation of Environmental Investments Research Program (EEIRP) work unit for the U.S. Army Corps of Engineers Institute of Water Resources (Thom and Wellman, in press). This guidance recommends the use of either conceptual or numerical models for developing linkages between goals and parameters that can be used to assess performance. Thom and Wellman (in press) define the process for developing performance criteria; this report takes the next logical step in identifying parameters that are appropriate to the variety of aquatic systems potentially considered for restoration.

UNCERTAINTY IN RESTORATION PROJECTS

Successful restoration of ecosystems is uncertain, and management of the restored system requires a continuous source of information. A monitoring program reduces uncertainty and forms the cornerstone of the assessment of the progress of the system. Guidance for developing a monitoring program for restoration projects has been developed for the Institute for Water Resources (Thom and Wellman, in press). This guidance illustrates how the monitoring program can develop efficient, cost-effective, and meaningful information about the system and explains how adaptive management benefits from monitoring.

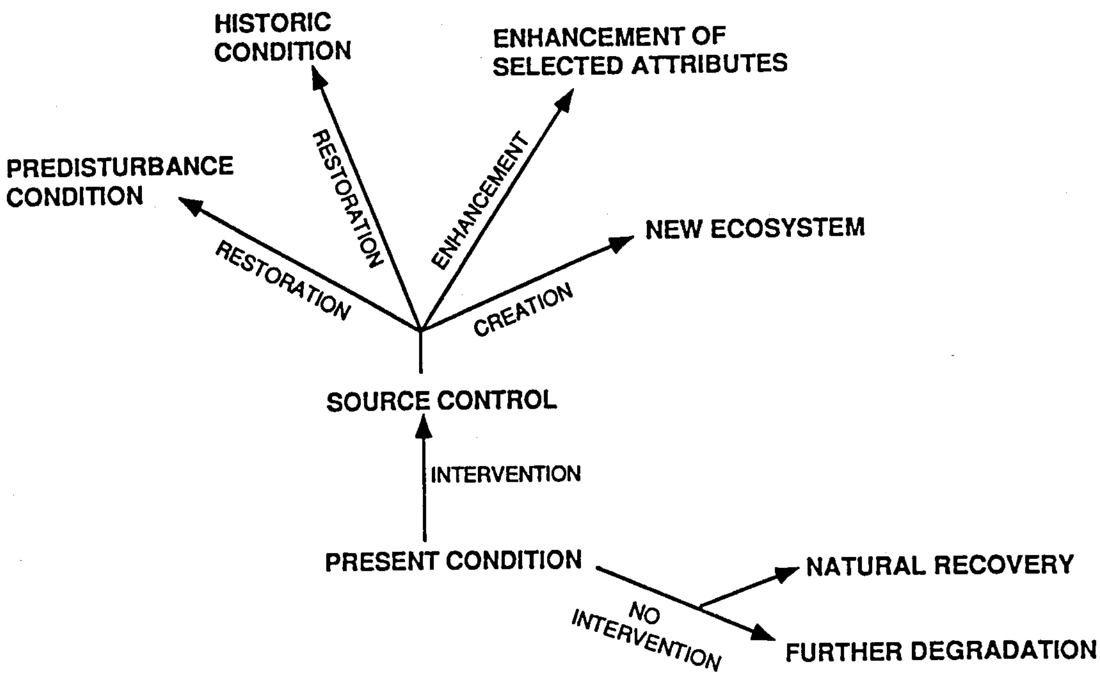
In a review of restoration strategies, Shreffler and Thom (1993) synthesized information from a variety of sources on the directions that a restoration project can take (Figure 4-1). Besides the option of no action, restoration can be accomplished through source control and natural recovery, restoration to predisturbance (i.e., before man) and historic conditions (i.e., man present but little influence), enhancement of selected attributes (e.g., habitat types), and creation of new ecosystems (i.e., systems that previously did not exist at the site). The degree of uncertainty about the success of the restoration effort varies with the restoration strategy (Table 4-1).

NRC (1992) showed that probability of success varied also with degree of previous disturbance of the landscape and restoration site (Figure 4-2). The restoration strategies that are best suited to various disturbance situations are shown. Sites within landscapes that are relatively undisturbed will have the highest probability of restoration, with a good chance of being restored to a predisturbance condition.

ADAPTIVE MANAGEMENT IN RESTORATION

The role of adaptive management in restoration projects, as described in Chapter 2, is to both guide maintenance or modification of the project at hand and collect information useful for further restoration efforts. Walters (1986) has outlined three ways to structure adaptive management: 1) evolutionary or "trial and error"; 2) passive adaptive; and 3) active adaptive. Under the evolutionary method, early choices are made in a haphazard manner and later choices are made from a subset of choices that may give more desirable results (Walters and Holling 1990). Passive adaptive management is employed when, using the best available information, a single response model is selected and decisions are made based on this model. It is assumed, not always rightly so, that this model is correct. Finally, active adaptive management means that manipulations may be carried out to evaluate which model is best for the sake of both developing this understanding as well as enhancing the performance of the system. Most theoretical literature on resource management uses the passive adaptive approach. The active approach is most often applied in studies such as agricultural field tests.

In a restoration project, the active adaptive method may provide the most meaningful information for making decisions that will ensure the ultimate success



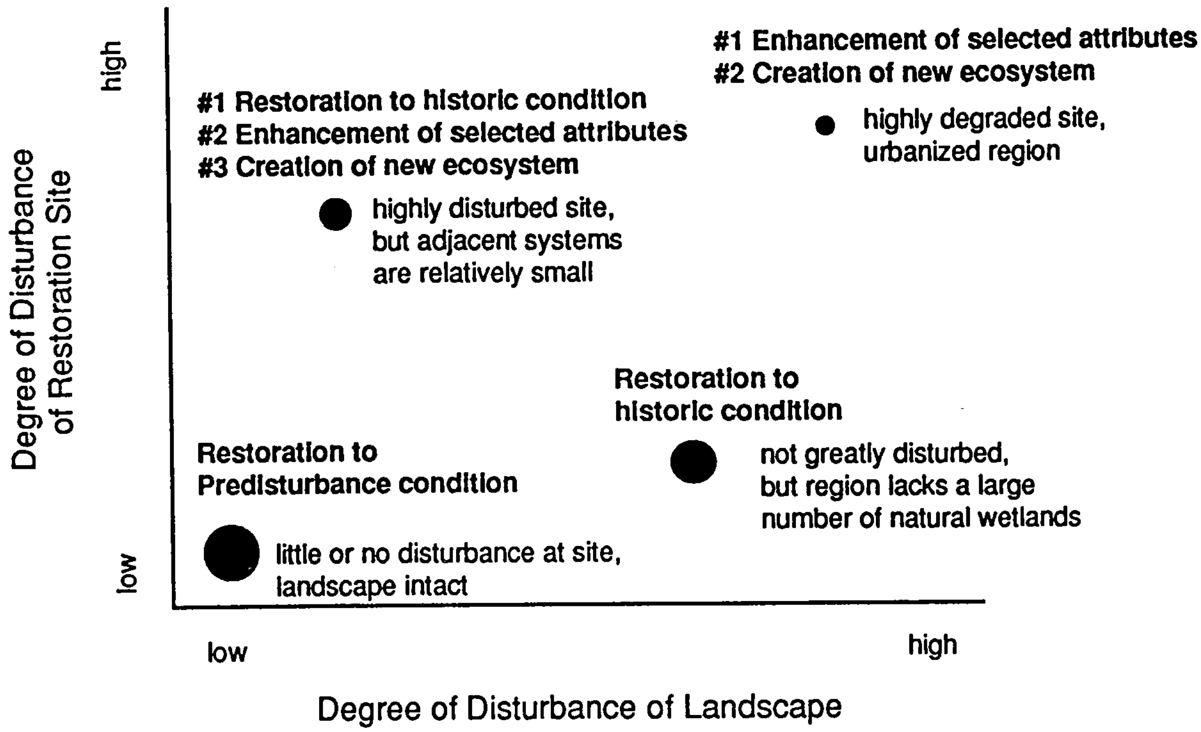
Source: Shreffler and Thom (1993)

FIGURE 4-1. Approaches to restoration.

TABLE 4-1. RANKING OF DIFFERENT APPROACHES FOR RESTORATION
(Best approach is 1, worst is 5)

Evaluation Criteria	Restoration to Predisturbance Condition	Restoration to Historic Condition	Enhancement of Selected Attributes	Creation of New Ecosystem	No Intervention
Predictability of Success	3	2	1	4	5
Short-Term Expediency	4	3	1	2	5
Long-Term Self-Maintenance	1	2	3	4	5
Habitat Composition in Landscape	1	2	3	4	5
Reduces Fragmentation	1	2	3	4	5
Benefits to Resources	1	2	3	4	5
Optimizes in View of Physical Modifications	4	1	2	3	5
Dependency on Water Quality	1	2	3	4	5
Does Not Alter Viable Habitat	4	3	2	5	1
Scientific Support:					
Data	4	3	1	2	5
Theory	1	2	5	4	3
Provides Benefits					
Outside Estuary	1	2	4	3	5

Source: Shreffler and Thom (1993).



Source: Shreffler and Thom (1993), as modified from NRC (1992)

FIGURE 4-2. Restoration potential and most appropriate approach for systems with varying degree of disturbance.

of the project and provide meaningful data that will help in the design of future projects. This method may also be the most costly and potentially harmful to the system, depending on the size and complexity of the experiments conducted. Walters and Holling (1990) note that designing experiments is not a trivial activity when managers require statistically significant results.

On a national level, NRC (1992) recommends that individual aquatic restoration projects be designed and executed according to the principles of adaptive planning and management. They suggest that plans be modified as technical knowledge improves and social preferences change. Under adaptive management, the knowledge gained through monitoring of the project and social policies must be translated into restoration policy and program redesign. The U.S. Environmental Protection Agency Wetland Research Program uses adaptive management as their strategy for improving design guidelines for wetland restoration projects (Kentula et al. 1992). The Wetland Research Project has been compiling a database on restoration projects and conducting directed research to refine the guidelines for restoration projects.

At least five major programs have recommended the use of adaptive management for restoration of damaged ecosystems. They include the following:

- The Forest Ecosystem Management Assessment Team recommended adaptive management as a critical element in the management and restoration of forest ecosystems in Pacific Northwest (FEMAT 1993). Planning, action, monitoring, evaluation, and adjustment were included in their process. Goals are revised based on monitoring, new knowledge, inventories, research, and new technologies.

- Boesch et al. (1994) reviewed wetland loss and restoration in Louisiana and recommended the use of adaptive management in the evaluation of restoration projects in that region. The adaptive management approach, coupled with effective monitoring, will provide a method to reduce the number of failed projects through providing cause and effect input to the management process.

- Restoration of the Kissimmee River, Florida, has specified adaptive management as the final of five parts of a comprehensive, ecological evaluation program designed to "...measure the pulse of the restoration project" (Toth 1995). In this program, adaptive management will provide for continual, scientifically informed, fine-tuning of the project and associated land and water management components.

- The restoration of Chesapeake Bay has followed an adaptive management approach in developing restoration actions, assessing performance, and adjusting the program (Hennessey 1994). An important lesson derived from the program is that human use of large estuarine systems creates a complexity that constrains management in a synoptic and comprehensive manner. The program has successfully used the information gathered and experiments conducted as a integrated learning tool to refine the program objectives and implementation actions.

- Gosselink et al. (1990) recommended that management and restoration of the extensive but diminishing bottomland hardwood forests in the south be conducted in an iterative approach. This approach involves ecological assessment, goal-setting, and planning for discrete implementation. They state that this process makes a provision for "institutional memory" so that actions are recorded as they occur and implementation strategies can be altered as the system approaches its goals.

Finally, the U.S. Department of Defense recommends that adaptive management be considered for inclusion in restoration projects that may have the potential for uncertainty in achieving restoration objectives (Corps 1995).

IMPLEMENTING ADAPTIVE MANAGEMENT

Annual Assessments

Adaptive management follows much the same guidelines as a physician diagnosing an illness: 1) the patient is evaluated (monitored); 2) the information is assessed against the body of knowledge about the symptoms; and 3) a remedy is prescribed. Under this scheme, ecosystem monitoring is at the heart of adaptive management (Corps 1995). Monitoring is conducted to evaluate the effectiveness of the remedy and to prescribe new remedies if needed.

Monitoring should be conducted on a frequency and for a duration most appropriate for the project (NRC 1990; Kentula et al. 1992). The frequency can range from very short (e.g., every second) to much longer (e.g., once every 2 years), depending on the parameter being measured and the questions asked. Regardless of the frequencies used for monitoring, annual assessments of the progress of the system should be made. The annual assessments would consider all relevant information, including monitoring data and additional data or expertise from outside the project. Based on this annual review, decisions can be made regarding mid-course corrections or other alternative (contingency) actions. Because the overall goal is to make the project "work," while not expending large amounts of funds to adhere to inflexible and unrealistic goals, decisions would be made as to what combination of physical actions may be needed vs. alterations in project goals. Strict adherence to a inflexible goal sets the project up for failure.

System-Development Matrix

General Model

A system development matrix, as shown in Figure 4-3, is an example of how the restored system can be viewed during development. The matrix can also provide clues as to the problems encountered in system development and direction on how to rectify the problems. Adaptive management can be defined as the best action for a system that is developed not *a priori*, but through a sequential assessment of system states and dynamic relationships among elements in the system (Walters and Hilborn 1978).

The restored system will exist in a number of states though time. These states can be defined in terms of structural and/or functional attributes or parameters. In Figure 4-3, structural and functional attributes are displayed in three states. At the lower left, the system is essentially newly restored. At the upper right corner, the system is fully developed and functional. It can be assumed that the goal for a restored system is to move the system state from the lower left corner to the upper right corner as quickly as possible. In reality, the system may not move linearly between these two states, but may take alternate pathways or move back and forth between states. The reasons for alternative pathways of development may be related to factors intrinsic to the restoration site, external factors, initial conditions, or a combination of these reasons.

Although two, three, four, or more states are potentially defined, the three states used in the matrix are based generally on models of community succession (e.g., van der Valk 1981), plant strategies (e.g., Grime 1979) and disturbance-diversity (e.g., Sousa 1984). For example, under the disturbance-diversity model, low levels of disturbance will result in a community (or assemblage) that contains an intermediate number of species. Intermediate levels of disturbance will produce a community with a high number of species. Finally, at high levels of disturbance, diversity will decline to the lowest level. New systems may contain a few species to start with that are tolerant of nutrient-poor conditions in soils. Further development will result in a large number of species occupying the site. At later stages of development, species richness is intermediate and is dominated by longer-lived species that are adapted for competing for available space and nutrients (Grime 1979).

FUNCTION	Rudimentary	Intermediate	Climax
Optimal	•function is independent of structure •functions best at early stage of development	•functions best at intermediate stage of development	•function and structure fully developed •stable •self-maintaining •resilient to disruption
Intermediate	•functions intermediate at early stage •early in development •moderate disturbance/disruption	•functions intermediate at intermediate stage •intermediate stage of development •moderate disturbance/disruption	•moderate function at full structural development •moderate correlation of function with structure •moderate disturbance/disruption
Non or Low	•early in development •failed structure •high disturbance/disruption	•functions low at intermediate stage •incorrect community •moderate disturbance/disruption	•low function at full structural development •incorrect community

STRUCTURE

FIGURE 4-3. General system-development matrix.

Functions can follow the same pattern as diversity. For example, productivity is typically moderate in fully developed systems under low disturbance (Grime 1979). Moderate disturbance raises productivity to its maximum, and high levels of disturbance reduce productivity to very low levels (Sousa 1984).

Because the goals for most aquatic restoration projects center on structural and/or functional parameters indicative of a target desirable system condition and structure and function can fit into three states, it may be reasonable to apply the matrix to adaptive management of restored systems. The objectives and goals of the project are used to establish the axes and to define the desirable ultimate condition of the system.

There are five basic steps in applying the matrix. First, the ranges of values for each state of development can be inserted under each state on the x and y axes. Values indicating optimal state for the structural or functional parameter may be *less* than the intermediate state. Management goals may be directed at maintaining a function at an intermediate level (e.g., water quality improvement), which may mean that the structure of the system is less than optimal (i.e., not climax). Data from the literature and reference systems that closely mimic the desirable target condition provide the quantitative metrics for the axes.

The second step involves locating the position of the system in the appropriate box in the matrix. Here, data from the monitoring program and other relevant and appropriate information are used to locate the position of the site. Attributes placed on the axes can represent quantitative structural or functional parameters. In addition, qualitative (i.e., low, intermediate, optimal) and semi-quantitative measures can be used. Candidate qualitative measures include social values, economic benefits (can also be quantitative), and aesthetics. Semi-quantitative measures include habitat suitability indices developed through standard procedures such as the Habitat Evaluation Procedure (USFWS 1980). Landscape ecological indices may work well in the matrix also (Forman and Godron 1986). For example, assessment of the system using the interior to edge ratio vs. a measure of utilization of the system by edge species could be evaluated using the matrix. More than one matrix may be needed if there are more than two parameters being evaluated. Matrices developed using quantitative measures can also be used in conjunction with matrices developed using qualitative measures to more fully evaluate the condition of the site.

Third, the phrases in the boxes in the matrix are used to explain why the system is in this condition. These phrases also indicate potential actions to be taken to redirect system development. For example, it is found that the system state locates it in the lower right box in Figure 4-3. In this state, the system appears to have the desired climax structure, but it is functioning at a low level. This may mean that system function is normally low at climax or that the community is incorrect for the desired function. In the latter case, incorrect assumptions were initially made about the level of functioning associated with the target community.

In the fourth step, alternative solutions to rectify the condition of the system are listed and evaluated. The general alternative "solutions" to the above example may be to:

- Take no action in hope that the system may, through time, eventually meet functional performance criteria
- Take action to improve functioning
- Modify the functional performance objectives to better coincide with the structural conditions the system has developed.

Resolution as to the appropriate action may require conducting additional directed studies or further review of available information. The energy and costs of modifying the system to meet previously defined objectives may be great and beyond the availability of funds for the site. Hence, in the final step, any specific physical actions to redirect the progress of the system must be carefully considered in light of the potential for the site to be successfully modified and the cost for undertaking the action.

The matrix can be set up to accommodate the goals or desired condition of the project. For example, if the desired condition is to have a system that has optimal capacity to treat storm water runoff to improve water quality, the optimal functional and structural conditions required to meet this objective would define the upper right box. From this point in the matrix, the other state conditions can be developed and scaled.

The matrix also allows for modification of goals if deemed appropriate. If it is determined that the system is progressing acceptably, the values on the axes can be scaled as needed to represent the site specific conditions. This is often more realistic in view of the fact that reference sites are optimally only about 80 percent similar to restoration sites (Kentula et al. 1992).

The system development matrix forms a defensible entity that contrasts project objectives against information on performance for the restored system. Once established during the planning phase of the project, the matrix forms the hypotheses for the performance of the system. During the monitoring and management phase, it allows new information from outside the project to be evaluated. It also allows realistic adjustments in performance goals if needed. The matrix can be used as a focal point for discussions with other agency personnel and interested parties for managing the system. Finally, the matrix and relevant documentation can be stored in project files as concise documentation for decisions made regarding actions taken.

Benthic Community Example

The succession of benthic soft-bottom communities has long been the subject of investigation. In an analysis of colonization rates and patterns of benthic infauna on newly exposed dredged material in Long Island Sound, Rhoads et al. (1978) identified three successional groups in terms of species composition and species' production rates. The successional groups (i.e., a measure of structure) observed included early colonizers, which arrive early and may disappear early because of competition or predation; an intermediate group, which represents a transition between very early colonizers and species that persist for long periods of time; and colonizers that may appear early but maintain more or less constant population densities over long periods of time. Species production, a measure of function, for these groups was also summarized by Rhoads et al. (1978).

The successional groups for the Rhoads et al. (1978) model are shown in a matrix in Figure 4-4. The dominant species in each successional group and the ranges of their production rates are provided on the x and y axes, respectively. The typical development pattern, that is the one with the highest probability based on available information, moves from the lower left box to the upper right box in the figure. Variations in this pattern result in community states in other boxes. These variations, which vary in probability, can be explained by differences in the conditions of food, reproduction, or recruitment. For example, if the late successional group has a very high production, this may mean that food availability is very high. This condition has been documented in climax communities where food supply is high in southern California and Puget Sound, Washington, by Word (1990). In addition, reproduction may be enhanced because of higher than expected temperatures. Hence, explanations can be made for variations in the typical pattern of development.

If in this latter example it is acceptable that the stage three assemblage exists in a state of enhanced productivity, then no action would be required. If however, this state is viewed as undesirable, then action to reduce the food supply or alter physical conditions would be recommended. Information from the monitoring program may show that the enhanced production is due to anomalous conditions and may be short-lived. If this is the case, then the community should assume a typical level of productivity after a short period of time. Further studies may be needed to evaluate the best alternative, and these studies can be designed based on the likely causes for the variation from typical development.

SPECIES PRODUCTION (g/sq.m/d)	*Streblospio benedicti* *Capitella capitata* *Ampelisca abdita* *Owenia fusiformis* *Mulinia lateralis*	*Nucula annulata* *Tellina agilis* *Pitar morrhuana*	*Nephtys incisa* *Ensus directus* *Nassarius trivittatus*
0.01 - 0.05	•very early in colonization •physical conditions poor for early colonizers •food availability poor	•physical conditions suboptimal for second group colonizers •food availability suboptimal for second group	•typical latest stage of development
0.04 - 0.12	•early in colonization •physical conditions suboptimal for early colonizers •food availability suboptimal	•typical intermediate stage of development	•somewhat enhanced food at latest stage •somewhat enhanced reproduction/growth because of physical (e.g., temperature) conditions
0.06 - 0.57	•typical early in development	•enhanced food at intermediate stage •enhanced reproduction/growth because of physical (e.g., temperature) conditions	•highly enhanced food at latest stage •highly enhanced reproduction/growth because of physical (e.g., temperature) conditions

DOMINANT SPECIES IN SUCCESSIONAL GROUPS

FIGURE 4-4. System development matrix for benthic infauna colonizing dredged material in a marine system.

Disturbance plays a key role in structuring benthic communities, and Rhoads et al. (1978) examined the effects of repeated disposal of dredged material on succession. Essentially, dredged material disposal shifts the community from stage three to stage one or two (i.e., from the upper right box to the middle box and lower left box). Rhoads et al. (1978) showed that the production rates and dominant species in stages one and two may be better than those for stage three in terms of providing prey for fish in the region. Hence, if a goal of the project is to enhance fisheries resources, maintaining the benthic community in stage one or two may be a goal for the project.

Marsh Development Example

Unlike benthic infaunal communities in the example above, marsh-dominated systems generally present a more complex problem when evaluating succession under a restoration scenario. Typically, the structure of the system is monitored. However, performance criteria for the system are established for functions that may not be closely linked to these structural measures. For example, the goal of the Gog-Le-Hi-Te tidal wetland system in the Puyallup River estuary, Washington, was to support juvenile salmon, shorebirds, waterfowl, small mammals, and raptors (Simenstad and Thom 1996). The performance objectives included specific allocations of area for each of these groups (i.e., 50 percent of the area for juvenile salmon, 20 percent for waterfowl, and 10 percent each for shorebirds, small mammals, and raptors).

At the time of project planning (early 1980s), little was known regarding what specific habitats were best for each of these groups or what function of the restored system might be critical to each of the groups. Further complicating the planning process was the fact that absolutely no natural systems that could be used as a model or reference system for design or monitoring remained in the heavily industrialized and developed estuary. Knowledgeable scientists used the best available information to provide guidance on habitat types that should be included in the system to try to enhance the support for the target resource species. A 2.2-ha tidal wetland that included intertidal sedge, cattail, unvegetated mudflats, and tidal channels; 1.7 ha of upland grassland with a small freshwater marsh; and shrub and forested riparian habitats were designed and constructed. The basic premise was that the diverse system provided a landscape that accommodated all of the target species groups. To enhance the development of the sedge marsh, approximately 49,000 shoots of sedge were planted onto the intertidal flats.

Monitoring of the system between 1986 and 1993 showed that the sedge marsh rapidly developed to a maximum extent between 1986 and 1987, then declined to a very small area by 1993. Cattails grew to dominate much of the space originally occupied by sedge. In addition, more bare mudflat was exposed because of the loss of the marsh and channels rapidly filled because of heavy loads of sediment transported by the Puyallup River. The channels narrowed considerably during the first 7 years, and a new system of braided channels developed through natural hydrological processes. Hence, the system looked very different than was envisaged by the planners. The upland areas, which were largely in existence when the system was constructed, remain unchanged.

Monitoring of fish and birds showed that the system was used early on by these groups. Juvenile salmon were present in the system during the first year. Subsequent experimental studies proved that the fish were consuming prey resources in the system and were growing while resident in the system (Shreffler et al. 1990, 1992). Birds were observed in densities far greater than observed in other areas in the estuary and were feeding, resting, and, in some cases, reproducing and rearing in the system. Clearly the system was meeting the intended goals, but it had a much different structure than was originally designed.

Using a system development matrix for this project could have formalized the performance criteria for target resources. The structure of the habitat (i.e., sedge marsh) could have been placed along the x axis and the function of the habitat for target species such as juvenile salmon could have been placed on the y axis. For example, in an early stage, the sedge would have a low density and the densities of prey resource species for salmon would be within a range known to typify this stage of development. Next, the range of sedge densities and prey densities in fully developed sedge marshes could have defined the upper right box. The intermediate stage may have either been identified through field studies or experimental investigations. If an intermediate stage was not definable, then the matrix would be reduced to two states on each axis. In the real case, monitoring has indicated that the system exists in the upper left box in the matrix. That is, the sedge marsh was only poorly developed, but the system contained high densities of prey resources. According to the general model in Figure 4-3, the community was incorrect (i.e., did not meet predictions) but was meeting functional criteria. Further analysis showed that the poor development of sedge and the massive increase in cattail may have been due to variations in salinity and elevation. Based on this information, alternative actions can be developed. At present, however, agencies and the local sponsor (the Port of Tacoma) have not settled on any alternative actions for this system.

In the case of Gog-Le-Hi-Te, the functional performance of the system for juvenile salmonids is very complex. Prey production probably occurs in the marsh as well as in the channels and mudflat. In addition, prey are advected into the system from upstream sources. Trapped in the system, they provide a concentrated resource for salmon to feed on. Monitoring has shown this to be and thereby has served a primary function of adaptive management: *gain in* *knowledge*. Having this information has and will continue to help in the design and assessment of other systems. Modeling (both conceptual and numerical) can be a powerful tool in sorting out this complexity and defining optimal structural attributes to provide optimal functional performance.

REFERENCES

Boesch, D.F., M.N. Josselyn, A.J. Mehta, J.T. Morris, W.K. Nuttle, C.A. Simenstad, and D.J.P. Swift. 1994. Scientific assessment of coastal wetland loss, restoration and management in Louisiana. J. Coast. Res., Special Issue No. 20.

Corps. 1994. Central and southern Florida project reconnaissance report comprehensive review study. U.S. Army Corps of Engineers, Jacksonville District, FL. 237 pp.

Corps. 1995. Water resources policies and authorities ecosystem restoration in the Civil Works program. Engineer Circular 1105-2-210. U.S. Army Corps of Engineers, Washington, DC.

FEMAT. 1993. Forest ecosystem management: an ecological, economic, and social assessment. Forest Ecosystem Management Team, U.S. Department of Agriculture.

Forman, R.T.T., and M. Godron. 1986. Landscape ecology. John Wiley & Sons, New York, NY.

Gosselink, J.G., L.C. Lee, and T.A. Muir. 1990. The regulation and management of bottomland hardwood forest wetlands: implications of the EPA-sponsored workshops. pp. 638–671. In: Ecological Processes and Cumulative Impacts: Illustrated by Bottomland Hardwood Wetland Ecosystems. J.G. Gosselink, L.C. Lee, and T.A. Muir (eds). Lewis Publishers, Inc. Chelsea, MI. 708 pp.

Grime, J.P. 1979. Plant strategies and vegetation processes. John Wiley & Sons, New York, NY. 222 pp.

Hennessey, T.M. 1994. Governance and adaptive management for estuarine ecosystems: the case of Chesapeake Bay. Coast. Mgmt 22:119–145.

Kentula, M.E., R.P. Brooks, S.E. Gwin, C.C. Holland, A.D. Sherman, and J.C. Sifneos. 1992. An approach to improving decision making in wetland restoration and creation. EPA/600/R-92/150. A.J. Hairston (ed). U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR. 151 pp.

NRC. 1990 Managing troubled waters: the role of marine environmental monitoring. National Research council, National Academy Press, Washington, DC.

NRC. 1992. Restoration of aquatic ecosystems. National Resource Council, National Academy Press, Washington, DC. 552 pp.

Rhoads, D.C., P.L. McCall, and J.Y. Yingst. 1978. Disturbance and production on the estuarine seafloor. Am. Sci. 66:577-586.

Shreffler, D.K., C.A. Simenstad, and R.M. Thom. 1990. Temporary residence by juvenile salmon of a restored estuarine wetland. Can. J. Fish. Aquat. Sci. 47:2079-2084.

Shreffler, D.K., C.A. Simenstad, and R.M. Thom. 1992. Juvenile salmon foraging in a restored estuarine wetland. Estuaries 15:204-213.

Shreffler, D.K., and R.M. Thom. 1993. Restoration of urban estuaries; new approaches for site location and design. Prepared for Washington Department of Natural Resources, Olympia, WA. Battelle Pacific Northwest Laboratories. 107 pp.

Simenstad, C.A., and R.M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecol. Appl. 6:38-56.

Sousa, W.P. 1984. The role of disturbance in natural communities. Ann. Rev. Ecol. Syst. 15:353-391.

Thom, R.M., and K.F. Wellman. (In press). Aquatic restoration monitoring: guidance for planning, implementation and management of monitoring programs. Prepared for U.S. Army Corps of Engineers, Institute for Water Resources, Alexandria, VA.

Toth, L.A. 1995. Principles and guidelines for restoration of river/floodplain ecosystems-Kissimmee River, Florida. pp. 49-73. In: Rehabilitating Damaged Ecosystems. Second Edition. John Cairns, Jr. (ed). Lewis Publishers, New York, NY. 425 pp.

USFWS. 1980. Habitat evaluation procedures. ESM 102. U.S. Department of the Interior, Fish and Wildlife Service, Division of Ecological Services, Washington, DC.

van der Valk, A.G. 1981. Succession in wetlands: a Gleasonian approach. Ecology 62:688-696.

Walters, C.J. 1986. Adaptive management of renewable resources. McGraw-Hill, New York, NY.

Walters, C.J., and R. Hilborn. 1978. Ecological optimization and adaptive management. Ann. Rev. Ecol. Syst. 9:157–188.

Walters, C.J., and C.S. Holling. 1990. Large-scale management experiments and learning by doing. Ecology 71:2060–2068.

Word, J.Q. 1990. The infaunal trophic index, a functional approach to benthic community analysis. Dissertation, University of Washington, Seattle, WA. 237 pp.

5. ECOSYSTEM AND RESTORATION PROFILES

This chapter is divided into sections based on the following six ecosystem types:

- Open coastline and near coastal waters (Section 5A)
- Subtidal estuaries (Section 5B)
- Estuarine and coastal wetlands (Section 5C)
- Freshwater wetlands (Section 5D)
- Streams and rivers (Section 5E)
- Lakes and reservoirs (Section 5F).

Each section describes the ecosystem, key environmental processes, and the types of restoration projects that commonly occur within that ecosystem. Case studies are presented at the end of each section to illustrate various restoration techniques and approaches.

5A. OPEN COASTLINE AND NEAR COASTAL WATERS

David Gettleson

Open coastlines are stretches of coast that are not sheltered from wave action. Near coastal waters are defined herein as subtidal waters extending to about 1.6 km (1 mile) from shore. Other marine habitat types addressed in this document are subtidal estuarine habitats and estuarine and coastal wetland habitats. These habitats are discussed in subsequent ecosystem sections within the *Ecosystem and Restoration Profiles* chapter.

ECOSYSTEM PROFILE

Various intertidal and subtidal marine habitats occur along open coastlines and in near coastal waters. Intertidal habitats found along open coastlines can be either rocky shorelines or sandy beaches. Subtidal habitats found in near coastal waters include both hard bottom habitats (coral reefs, live bottom, worm rock reefs, artificial reefs, algal communities) and soft bottom habitats (seagrass beds, unvegetated soft bottom). By definition, open coastlines are high energy systems where tides and wave action play an important role. These ecosystems are naturally stressed systems with wide latitudinal ranges (Odum and Copeland 1974). The species present in these ecosystems differ substantially from coast to coast and along latitudinal gradients. Because the ecosystem characteristics of open coastlines are so varied and, thus, create sub-habitats, the following discussions of key ecological processes and ecosystem health are specific to the subhabitat type rather than the general coastal habitat.

Intertidal Habitats

Intertidal habitats along open coastlines are classified into two categories based on substrate: 1) rocky shorelines and 2) sandy beaches and dunes. These habitat types are discussed below.

Rocky Shorelines

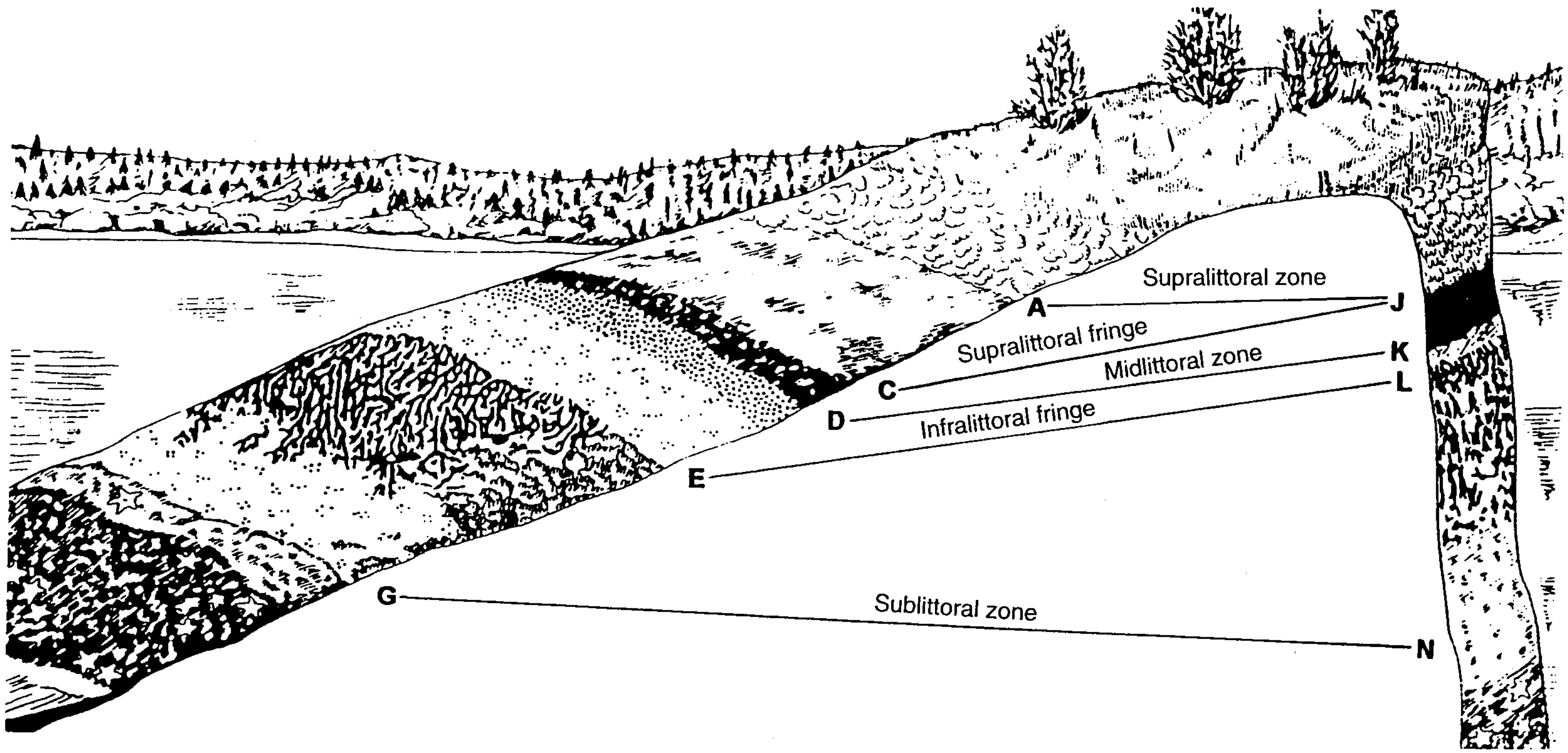
Geographic Distribution—Rocky shorelines occur along all coastlines of the United States but are more common in the north Atlantic and along the Pacific coasts than along the south Atlantic coast and the Gulf of Mexico (Odum et al. 1974; Nybakken 1988; Gross 1993).

Zonation Within Habitats—Intertidal habitats along rocky shorelines develop characteristic communities of attached plants and animals that live in bands or zones both above and below mean sea level (Lewis 1964; Stephenson and Stephenson 1972). Although community zonation patterns remain remarkably similar for rocky shorelines everywhere, the vertical extent of these zones depends on slope, tidal range, and exposure to wave action. The classical zonation pattern (Figure 5A-1) described for rocky shores consists of the following:

- The supralittoral or spray zone above the highest tides, characterized by salt-tolerant terrestrial vegetation
- The supralittoral fringe, characterized by littorine snails (*Littorina* sp.) and black encrusting lichens (*Verrucaria* sp.)
- The midlittoral zone, characterized by barnacles (*Balanus* sp.). Along the Pacific Northwest coast, mussels (*Mytilus californianus*) and predatory starfish (*Pisaster ochraceus*), as well as several species of predatory gastropods (*Thais* sp.), are present in this zone
- The infralittoral fringe, characterized by kelps (*Laminaria* sp.)
- The sublittoral or subtidal area, characterized by macroalgae.

Biological Community—Similar communities develop on rocky intertidal substrates in different areas, although individual species may differ depending on latitude, exposure to wave action, substrate characteristics, and other factors. Along northeastern United States rocky shorelines, dominant species, in terms of space occupied, are barnacles (*Semibalanus balanoides*), mussels (*Mytilus edulis*), and algae (*Ascophyllum* spp., *Chondrus crispus*, *Fucus* spp.) (Mathiesen et al. 1991). Barnacles usually dominate upper shorelines at exposed and moderately exposed sites. Depending on wave exposure and other factors, *Ascophyllum* spp. or *Fucus* spp. will usually dominate the mid-littoral zone, but in areas of extreme wave action, *Mytilus edulis* become the major occupier of space. Common grazers (herbivores) include various crustaceans and gastropods. Important predators include starfish (*Asterias vulgaris*) and whelks (*Thais lapillus*).

Dominant sessile species in Southern California rocky intertidal communities include encrusting blue green algae, coralline algae, and barnacles (*Chthamalus* spp.) in the upper intertidal zone; red algae (*Gigartina* spp.), mussels (*Mytilus californianus*), surfgrasses (*Phyllospadix* spp.), and sand-castle worms (*Phragmatopoma californica*) in the mid-intertidal zone; and surfgrasses and various red and brown algae in the low intertidal zone (Littler et al. 1991).



LEGEND

A-J Lower limits of lichens
C-J Upper limit of "black" zone
D-K Upper limit of barnacles
E-L Upper limit of *Fucus*
G-N Lower limit of barnacles

Source: From LIFE BETWEEN TIDEMARKS ON ROCKY SHORES by Stephenson and Stephenson. Copyright (c) 1972 by W.H. Freeman and Company. Used with permission.

FIGURE 5A-1. Classical zonation pattern for rocky shorelines.

Pacific Northwest rocky intertidal communities generally include a high splash zone with littorine snails, a barnacle (*Balanus* sp.) zone, a mussel (*Mytilus californianus*) zone, and a low surfgrass (*Phyllospadix* spp.) zone (Foster et al. 1991). The mid-littoral zone is spatially dominated by mussels. The predatory starfish (*Pisaster ochraceus*) preferentially preys on *M. californianus* and keeps it from overgrowing barnacles in some areas (Nybakken 1988).

Key Ecological Processes—Intertidal hard substrates are important producing, consuming, and recycling components of the nearshore marine ecosystem (Odum and Copeland 1974). Tides, wave action, light, temperature, and desiccation are major physical factors affecting species composition and zonation of rocky shorelines. Competition, predation, grazing, and succession are important biological processes. Physical factors may be more important in the upper intertidal zone, whereas biological processes may be more important in the lower intertidal zone (Nybakken 1988).

Nutrient Sources and Distribution: Nutrient sources for rocky intertidal communities include surface runoff, detritus in the water column, and photosynthesis by the algal species occurring there. Wave action and water movement distributes nutrients to the primary producers and detrital or filter feeding organisms.

Detrital Processing and Nutrient Regeneration: Mussels and other filter feeders collect and mineralize plankton and detritus, releasing inorganic nutrients to the algal species present.

Habitat Heterogeneity: Substrate heterogeneity is a basic feature of rocky intertidal habitats that contributes to the diversity of the associated community. Variations in substrate type, slope, exposure, and rugosity (roughness) create environmental patchiness that is superimposed on the broad zonation scheme discussed above. A variety of organisms have adapted to make use of crevices, overhangs, tidepools, and rocks of different slope and orientation throughout the rocky shoreline.

Key Natural Disturbances: Wave action (especially during storms) is a frequent source of disturbance for rocky intertidal communities, producing much of the small-scale patchiness. Other natural disturbances include unusually large rainfall events (which produce low salinities) and periods of prolonged exposure. Ice is also a major physical factor at higher latitudes in the winter months. Predation and grazing are important biological disturbances in rocky intertidal communities. When organisms are removed from patches of substrate by any of these mechanisms, a succession pattern is seen while the substrate is repopulated. Opportunistic species, such as filamentous and mat forming algae and hydroids, are usually among the first colonizers, followed by more advanced algal species, carnivores, and attached animals. Continuing disturbances ensure that some kind of succession is always occurring in some parts of a rocky intertidal community.

Landscape Interactions: Rocky shorelines recycle nutrients and interact with the surrounding benthic and nektonic communities as both a consumer of detrital material and a primary producer. Mussels and other filter feeders help make nutrients available to organisms within the water column.

Functional Values—Rocky shorelines naturally protect continental uplands from erosion. They also provide habitat and/or feeding grounds for many species of marine birds and marine mammals, such as seals, sea otters, sea lions, and walrus.

Causes for Deterioration—Rocky shorelines may be affected by modifications in water flow patterns, tidal range, and water quality (e.g., contamination by pollutants). Erosion and other mechanical habitat destruction can also cause loss of habitat.

Assessment of Habitat Health—Potential indicators of habitat health in rocky intertidal communities include contaminant concentrations in sessile organisms (e.g., mussels) at sufficient levels to cause population effects on the organism or food chain effects to humans or other predators.

Sandy Beaches and Sand Dunes

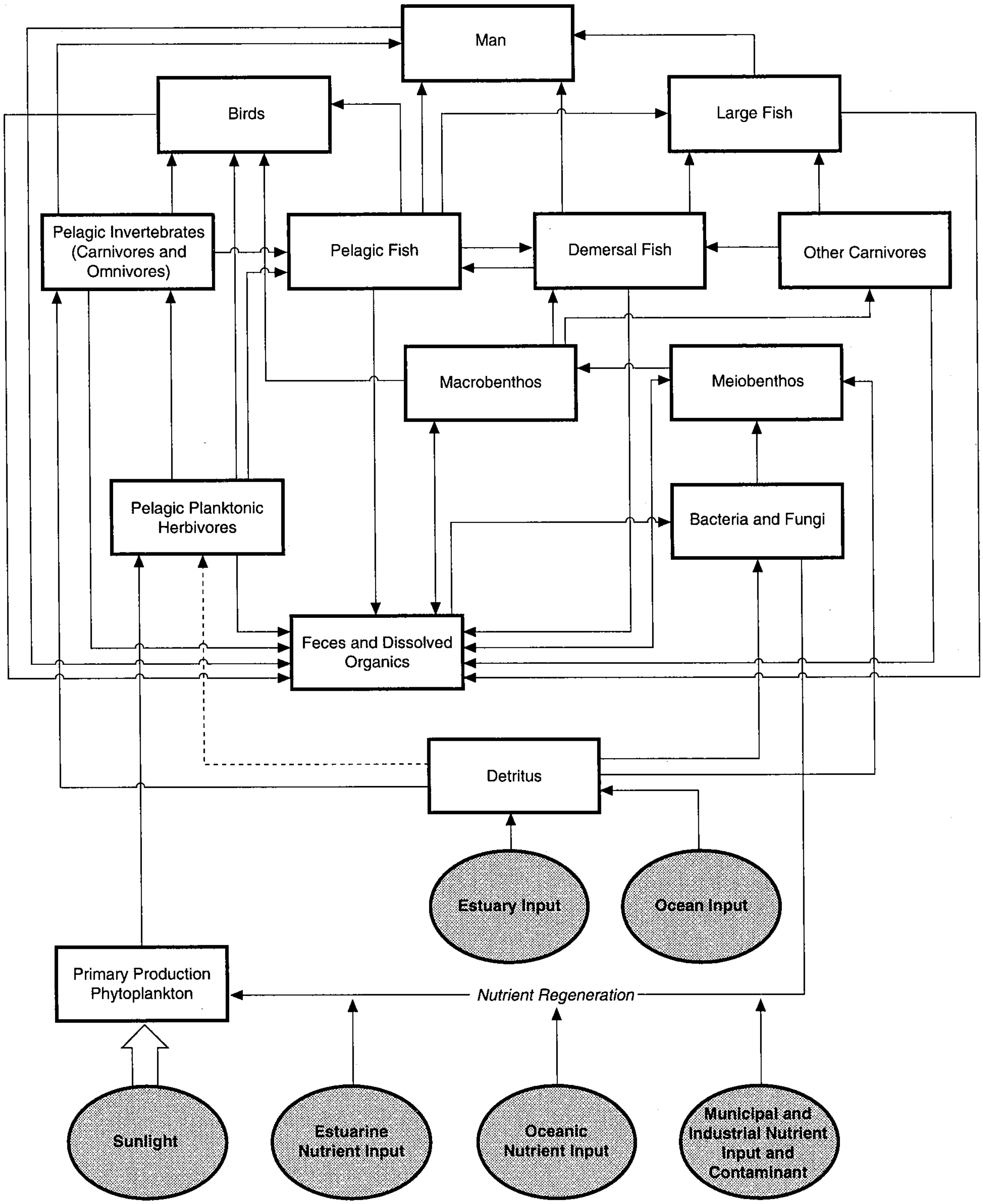
Geographic Distribution—Sandy beaches and sand dunes occur along all coastlines of the United States, but they are more common along the mid- and south Atlantic coast and along the Gulf of Mexico than along the north Atlantic and Pacific coasts.

Zonation Within Habitats—Above the "low water mark," beaches are divided into the foreshore, which contains the "low tide terrace" or area covered by water at high tide, and the beach face or scarp, which slopes down from the sand dune or berm to the "high water mark." Behind the beach face, sand, wind, and vegetation interact to form the coastal dune community. The foreshore ends at the crest of the berm and gives way to the back shore, which may contain other sand dunes or berms (Komar 1976; Gross 1993). The biological communities described below reflect these physical zones.

Biological Community—The unstable nature of the substratum on sandy beaches excludes most large epifauna. Filter-feeding species of burrowing bivalves, polychaetes, and crustaceans are the dominant macrofauna. High intertidal levels are typically occupied by burrowing amphipods, which emerge at night to feed on beach wrack. Other scavengers include shorebirds, fishes, ghost crabs, and insects. Diatoms, ciliates, copepods, gastrotrichs, gnathstomulids, tardigrades, and turbellarians live in the interstices between the sand grains. With their extended and complex faunal communities, beaches form an extensive food filtering system characterized by a variety of suspension feeders, deposit feeders, and scavengers. Multiple food webs are present and are based on detritus, dissolved nutrients, plankton, and other plants and animals (Figure 5A-2). In addition, along the southeastern United States and throughout the Gulf of Mexico, sandy beaches are critical nesting habitat for several species of endangered and threatened sea turtles (Pritchard 1979; Sandifer et al. 1980).

Dunes represent the sand reserves for the beach (Kaufman and Pilkey 1979). Sand dunes are stabilized temporarily by a variety of salt-spray-adapted vegetation, which includes beach hogwort (*Croton punctatus*), sea oats (*Uniola paniculata*), Russian thistle (*Salsola kali*), seabeach orach (*Atriplex arenaria*), sea rocket (*Cakile harperi*), seabeach panic grass (*Panicum amarum*), and creeping spurge (*Euphorbia serpens*). Sea oats and seabeach panic grass are particularly important in the stabilization and growth of dunes because both species tolerate intensive salt spray and develop an extensive lateral root system. Vegetation in the beach dune community provides habitat for a large number of terrestrial species, including snakes, lizards, tortoises, birds, and small mammals. Many of these species are restricted in distribution because coastal development has severely reduced their available dune habitat. As a result, disproportionately large numbers of species from the beach dune animal community are currently listed as endangered, threatened, or species of special concern (Sandifer et al. 1980; Wood 1989).

Key Ecological Processes—Sandy beaches are high energy habitats where strong wave action, currents, and the daily ebb and flow of tides move sand and sort it into zones of coarser and finer particles. Therefore, sandy beaches are essentially sediment deposits "on the move." The most important physical factor governing life on sandy beaches is wave action and its effect on particle size. Wave action is responsible for substrate movement, which controls all community



Source: Modified from Steele (1974).

FIGURE 5A-2. A generalized food web for the coastal marine ecosystem in the mid-Atlantic coastal region.

development on the foreshore and berm of sandy beaches. Wind and rainfall play an important role in dune community development.

Nutrient Sources and Distribution: Primary productivity on sandy beaches is very low. There are no large, multicellular plants below the high tide line, and microalgae, such as diatoms, are restricted to near-surface layers because light does not penetrate very far into sand. Therefore, animals living on sandy beaches depend on phytoplankton or organic detritus brought in by wave action. In sand dune areas, plant and animal communities show nutrient source and distribution patterns typical of most terrestrial ecosystems.

Detrital Processing and Nutrient Regeneration: Because there is little primary productivity within sandy beaches, little plant detritus originates from within the system. However, beach wrack and other types of detritus from adjacent waters are very important to the system. Scavenging activities by macrofauna, such as amphipods and isopods, may accelerate the physical and microbial breakdown of detritus particles.

Habitat Heterogeneity: There is little habitat heterogeneity within sandy beaches. Compared with rocky shorelines, these habitats show a much smoother, more uniform profile.

Key Natural Disturbances: Sediments on sandy beaches are constantly being moved and sorted by waves and wind. The impacts of storms vary with the intensity and frequency of storm events; however, storms generally accelerate the long-term erosion process both along beaches and within the beach dune system.

Landscape Interactions: Sandy beaches and their associated sand dune areas interact with both the open ocean on the beach side and the terrestrial or estuarine ecosystems inshore of them. The beach dune system is essentially a transitional habitat between the terrestrial and marine ecosystems. On a landscape scale, beach sediments are constantly being moved in the longshore drift.

Functional Values—Sandy beaches and sand dune systems are sand reserves that prevent or delay erosion of inshore terrestrial habitats and property. They represent immense economic value to humans in terms of erosion prevention, tourism, recreation, and aesthetics. Ecologically, they provide critical habitat for many species of threatened or endangered species, such as sea turtles, small mammals, and various ground-nesting birds.

Causes for Deterioration—Erosion is the main mechanism of habitat loss from sandy beaches and is usually the reason for conducting a restoration project. Erosion may be the result of natural processes, such as storms, or it may be induced by channel and inlet maintenance and the presence of structures such as inlets and jetties.

Assessment of Habitat Health—Potential indicators of habitat health in sandy beaches include infaunal abundance and species composition and concentrations of contaminants in sediment. Dune vegetation surveys are useful in determining the health of sand dune systems.

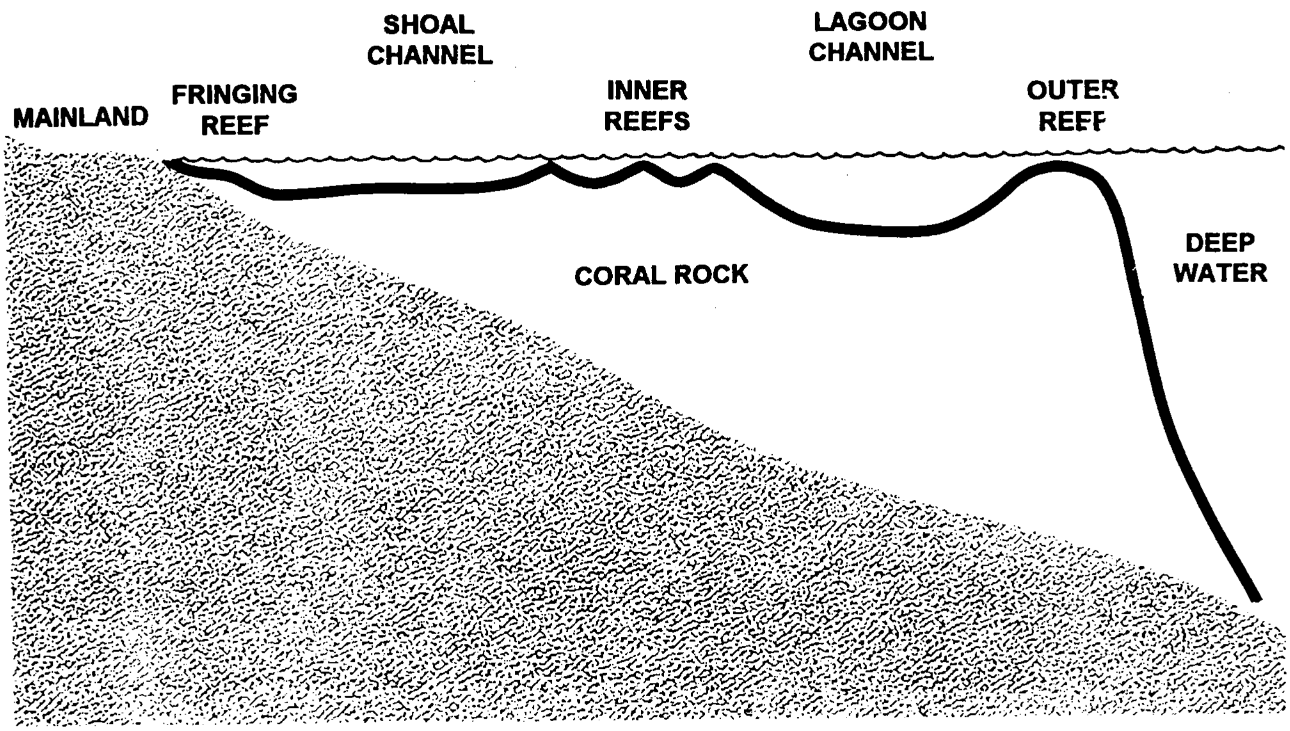
Subtidal Habitats

Subtidal habitats are generally characterized, based on their substrate, as either hard bottom habitats or soft bottom habitats. Hard bottom habitats include coral reefs, live bottom areas (i.e., low relief hard bottom), worm rock reefs, artificial reefs, and algal communities. Soft bottom habitats include seagrass beds and non-vegetated soft bottom communities. These habitat types are described below.

Coral Reefs

Geographic Distribution—Although coral reefs cover millions of square miles within tropical waters, the only actively growing nearshore coral reefs along the continental United States are located off the Florida Keys (Jaap 1984; Shinn et al. 1989).

Zonation Within Habitats—Coral reefs are large, complex species associations that include several habitats and microhabitats. Reef development and species zonation within particular reefs depends to a great extent on geographic location. It is difficult to give a generally accepted pattern of reef zonation applicable to all reefs (Smith 1972; Nybakken 1988). Figure 5A-3 illustrates the type of reef formation seen off the Florida Keys. The Florida reef tract is composed of more than 96 km (60 miles) of outer bank reefs and more than 6,000 inner and fringing patch reefs (Marszalek et al. 1977). Table 5A-1 shows coral species zonation with depth patterns that are typical of the outer reefs along the Florida reef tract.



Source: Modified from Smith (1972).

FIGURE 5A-3. Generalized structure of coral reefs seen off the Florida Keys.

TABLE 5A-1. A BANK REEF ZONATION PATTERN TYPICAL OF THE SOUTH FLORIDA REEF TRACT

Zone	Depth (m)	Conspicuous organisms
Back reef/rubble area	0.6–1.8	*Porites astreoides, Favia fragum*
Reef flat	0.6–1.2	*Diploria clivosa, Porites astreoides,* crustose coralline algae
Shallow spur and groove	1.2–2.4	*Millepora complanata, Palythoa* sp.
Deep spur and groove	2.4–4.6	*Gorgonia ventalina, Acropora palmata*
Buttress or fore reef	4.6–30.0	*Montastrea annularis, Diploria strigosa, Colpophyllia natans*
Deep reef	41.1	*Helioseris cucullata, Agaricia fragilis, Madracis mirabilis*

Source: Jaap (1984).

Biological Communities—Coral reefs represent a tropical shallow water ecosystem that is extremely productive. Reef-building algae and corals remove calcium ions from seawater and incorporate them into their skeletons, that in turn form the limestone base of the reef. This limestone substratum provides habitat for a vast array of organisms that attach to it, crawl over it, hide within it, or swim near it. The principal builders of the coral reef habitat are the hermatypic or hard corals. Other important contributors to the limestone substratum of a coral reef are the crustose, calcareous red algae, and some species of green algae that produce limestone skeletons.

Coral reef ecosystems are characterized by high species diversity, rapid recycling of nitrogen and phosphorus, high gross primary productivity, and low net primary productivity resulting from high respiration rates (Jaap and Hallock 1990). Food webs are extremely complex, and many species present have specialized food requirements, narrow niches, and complex life cycles. Symbiotic relationships are common, and a large amount of primary productivity happens within the zooxanthellae (microscopic symbiotic algae) living inside the tissue of the reef-dwelling hard corals (Goreau and Goreau 1960; Porter et al. 1984). Table 5A-2 presents a list of the more common plants and animals seen within the Florida reef tract ecosystem.

Key Ecological Processes—Coral reefs develop on a limestone substratum and require bright light, stable high salinity, and temperatures of 21°C (70°F) or higher. Although there are many species of coral and algae that grow and calcify in deep, cold waters, true coral reef development is rarely seen at depths greater than 100 m (330 ft) or at temperatures below 18°C (64°F) or above 36°C (97°F). Reef development also requires clear, oligotrophic (nutrient poor) waters. Important biological processes within coral reef systems include predation, grazing, and interspecific competition.

Nutrient Sources and Distribution: Coral reefs rapidly recycle nitrogen and phosphorus within the community. Reefs have high gross primary productivity because of their algal component, but their overall net primary productivity is low because of the high respiration rates of the organisms within the community.

Detrital Processing and Nutrient Regeneration: Reefs grow in nutrient poor water and have the ability to tenaciously hold onto and recycle all nutrients within the system. Reefs also act as a sink for nutrients entering from the outside.

TABLE 5A-2. COMMON PLANTS AND ANIMALS SEEN ALONG THE FLORIDA REEF TRACT

Algae
Goniolithon strictum
Halimeda opuntia
Halimeda tuna
Penicillus sp.
Porolithon sp.

Angiosperms
Thalassia testudinum

Sponges
Numerous, but not recorded

Corals
Acropora palmata
Acropora cervicornis
Montastrea annularis
Montastrea cavernosa
Porites astreoides
Porites porites
Siderastrea radians
Siderastrea siderea
Diploria strigosa
Diploria clivosa
Diploria labyrinthiformis
Agaricia agaricites
Millepora complanata
Mycetophyllia lamarckana
Isophyllia multiflora
Eusmilia fastigiata
Dichocoenia stokesii
Palythoa mammilatus
Gorgonia flabellum
Eunicea sp.
Plexaurella sp.
Pseudopterogorgia sp.

Molluscs
Lithophaga antillarum
Arca sp.
many other bivalves
Strombus gigas
Astraea caelata
Vasum muricatum
many other gastropods

Arthropods
Panulirus argus
several others

Echinoderms
Diadema antillarum
Eucidaris tribuloides
several others

Fishes
Sphyraena barracuda (barracuda)
Chaetodon sp. (butterfly fish)
Anisotremus virginicus (porkfish)
Lutjanus grisseus (mangrove snapper)
Ocyurus chrysurus (yellowtail snapper)
Scarus coeruleus (blue parrotfish)
Abudefduf saxatilis (sergeant major)
Thalassoma bifasciatum (bluehead wrasse)
Lactophys quadricornis (scrawled cowfish)
Acanthurus sp. (surgeonfishes)
Pomocanthus arcuatus (gray angelfish)
Pomocanthus paru (French angelfish)
Epinephelus itajara (jewfish)
Haemulidae (grunts)
Strongylura marina (Atlantic needlefish)
Epinephelus nigritus (Warsaw grouper)
Selene volmer (lookdown)
Ogcocephalidae (batfishes)
Equetus lanceolatus (jackknife fish)
Acanthurus coeruleus (blue tang)
Chaetodipterus faber (spadefish)
Balistes vetula (queen triggerfish)
Muraenidae (morays)
Aetobatus narinari (spotted eagle ray)
and many others

Other
Numerous encrusting bryozoans and foraminifers

Source: Multer (1969).

Habitat Heterogeneity: Extensive habitat heterogeneity is seen within a coral reef community. Macrohabitat variation is seen between "back reef," "rubble zone," "reef crest," and "fore reef slope" portions of the reef. Microhabitat variation exists within the structure of the massive corals themselves, under overhangs, along ledges, and in shaded areas.

Key Natural Disturbances: Storms, predation, and disease are the major natural disturbances seen in coral reefs. Storms are infrequent, but when they occur, they have a dramatic impact on the reef crest and shallow reef zones. Disease and predation are continually occurring within coral reefs. Under normal circumstances, these destructive processes are balanced by reef building processes. Infrequently, however, there are outbreaks of disease or population explosions in some specific predator that may have very destructive impacts in localized reef areas.

Landscape Interactions: Coral reefs in the Florida Keys are part of a complex, landscape mosaic that includes seagrass beds and mangrove forests. Energy, chemical constituents, and fish and motile invertebrate species are exchanged between these systems. Coral reefs provide shelter for organisms such as fishes and lobsters that hunt at night in adjacent seagrass beds. Mangroves and seagrass beds intercept nutrients and sediments entering coastal waters from terrestrial sources, thereby helping to maintain the clear water needed for reef development. With their more plentiful supplies of nutrients, mangrove and seagrass communities also provide nursery areas for many juvenile fishes and invertebrates that later inhabit the reefs. Finally, bank reefs, which form barriers at the oceanic margins, create sheltered lagoons behind which seagrass beds can develop.

Functional Value—As the only living examples of coral reefs within the nearshore coastal waters of the United States, the reefs in the Florida Keys are an extremely valuable natural resource. Their importance to the nation as a whole has recently been recognized by the designation of the Florida Keys National Marine Sanctuary. Within the Florida Keys, the Florida coral reefs are the foundation of a multimillion dollar tourist and recreational diving industry.

Causes for Deterioration—Coral reefs are very sensitive to water quality conditions, such as elevated turbidity and nutrient concentrations, which can result from human activities onshore (e.g., sewage disposal, increased sediment runoff in stormwater caused by construction activities). Reefs can also be damaged by mechanical mishaps such as accidental ship groundings and anchoring. Complicating their existence are potentially catastrophic natural processes such as outbreaks of predators and disease.

Assessment of Habitat Health—Potential indicators of habitat health in coral reef systems include water clarity, nutrient concentrations and ratios (in water, sediment, and tissue), macroalgal abundance, incidence of bleaching, incidence of disease, and abundance of major grazers (e.g., sea urchins [*Diadema* sp.]).

Live Bottom Areas

Subtidal hard bottom areas colonized by sessile and motile epibiota are often referred to as "hard grounds" or "live bottom" areas. Unlike coral reefs, where corals are building the hard substrate upon which the community grows, live bottom communities exist on submerged rocks or hard fossil substrates (e.g., former reefs drowned by sea level rise). Algae are often present in these live bottom areas to varying degrees; subtidal hard bottom areas dominated by algal communities (e.g., kelp beds) are discussed separately below.

Geographic Distribution—Live bottom areas occur on all continental shelves in the United States. However, they are most common on the west Florida shelf and along the north and south Atlantic coast (Parker et al. 1983; CSA 1989). Live bottom areas also occur throughout the northern Gulf of Mexico to the Texas/Mexico border (Jaap 1983) and at various locations off the Pacific coast.

Zonation Within Habitats—There is no single recognized zonation pattern for live bottom areas, although differences in community composition with vertical relief, depth, and distance from shore have been recognized (Rezak et al. 1985). Surf zone and very nearshore live bottom communities are characterized by low-relief boring or encrusting organisms (Jackson 1979; Taylor 1979). Species richness, as well as vertical relief of the sessile invertebrates and macroalgae growing on hard substrates, increases with distance from shore.

Biological Community—Live bottom communities develop on subtidal hard bottom outcrops and become aggregation centers for fish and invertebrate species. Community composition varies so much, both geographically and among outcrops in a given area, that it is not possible to list all the individual species that are typically found in live bottom areas. Therefore, only broad groups are discussed herein.

Sessile macroinvertebrates typically associated with hard bottom areas include sponges, hydroids, octocorals, ahermatypic scleractinian corals, bryozoans, and ascidians. Motile macroinvertebrates include various crustaceans, molluscs, polychaetes, and echinoderms. While macroalgal species may be important members of a live bottom community, they are usually not the dominant members in terms of cover. Fish assemblages may include a mixture of primary and secondary reef fishes (Smith 1976) that are attracted to the cover and food source provided by the hard substrates, as well as soft-bottom dwellers that inhabit sand patches between outcrops.

Key Ecological Processes—Unlike coral reefs, which have relatively narrow environmental tolerances, live bottom communities occur in a wider range of environmental conditions. Because the substrate usually has little vertical relief, sedimentation and sediment movement are critical factors, and many of the organisms have adapted to withstand sedimentation or to quickly colonize newly exposed patches of hard substrate. In nearshore areas, wave action is a major agent of change. Temperature, water clarity, and nutrient concentrations also affect species composition in live bottom communities.

Nutrient Sources and Distribution: Live bottom communities are generally characteristic of nutrient poor waters. The corals and invertebrates present rapidly recycle nitrogen and phosphorus. Net primary productivity is usually low, but some areas develop significant algal assemblages.

Detrital Processing and Nutrient Regeneration: Relatively little detrital reprocessing takes place in live bottom communities. Nutrients enter the community from coastal upwelling, from primary productivity within the water column, and from surrounding seagrass and algal communities.

Habitat Heterogeneity: Live bottom communities are heterogenous because of the structural complexity of the underlying hard substrate. Many hard bottom areas are patchy, consisting of outcrops, ledges, holes, and interspersed areas of sand bottom. Sessile epibiota add further to the structural complexity, providing a variety of microhabitats.

Key Natural Disturbances: Storms and inundation by sediments are the most common natural disturbances within live bottom communities. Wave surges associated with storms break off and dislocate attached members of the invertebrate fauna. Most live bottom communities have adapted to this type of periodic disruption and recover rapidly. Sand sheet migration associated with storms or currents can alternately uncover and bury areas of hard substrate, destroying epifaunal growth in some areas and providing new substrate for colonization in others.

Landscape Interactions: In many areas, hard bottom and soft bottom habitats are interspersed to create a landscape-level mosaic. Hard bottom areas provide shelter for various fishes and invertebrates that feed on macroinfauna in surrounding soft bottom areas. In addition, hard bottom areas provide important habitat for many species of coral reef fish and invertebrates, extending the ranges of these species far beyond the areas where coral reef communities actually exist.

Functional Values—Live bottom communities are recognized as centers for fish and invertebrate production along the continental shelf. The federal government and individual states consider them valuable environmental resources and protect these habitats when coastal projects are considered for permitting.

Causes for Deterioration—While not as sensitive as coral reefs, live bottom areas may be affected by water quality conditions, such as elevated turbidity and nutrient concentrations, which can result from human activities onshore. Live bottom areas may also be mechanically damaged by activities such as trawling, dredging, and anchoring. Natural processes, such as sediment movement, can also alternately expose new substrate and bury organisms.

Assessment of Habitat Health—Potential indicators of habitat health in live bottom communities include biotic cover and species composition; macroalgal abundance; water clarity; nutrient concentrations and ratios in water, sediment, and tissue; and concentrations of contaminants in sediment and tissue.

Worm Rock Reefs

Geographic Distribution—Within the continental United States, worm rock reefs occur only along the southeastern coast of Florida from the Cape Canaveral area through Key Biscayne (Kirtley 1966).

Zonation Within Habitats—Generally, worm rock reefs are seen near the intertidal zone or in the surf zone off sandy beaches. They also occur within the mouths of inlets and along rock jetties.

Biological Community—Worm rock reefs are constructed by colonies of the tube-building worm (*Phragmatopoma lapidosa*). These sabellariid polychaetes live in tubes they build around themselves by cementing sand grains together. Within their geographic range, tube-building worm colonies build large, geologically and biologically significant structures that can extend along hundreds of kilometers of coastline. Worm rock reefs also provide the habitat for an elaborate and stable community of many other marine species. They support a diverse assemblage of other invertebrate species and, in some cases, provide habitat for juvenile and cryptic fish species (Kirtley 1974; Gore et al. 1978; Gilmore et al. 1981; Van Montfrans 1981). Rudolph (1977) observed 88 species of other polychaete annelids living in association with worm rock reefs. Gore et al. (1978) and Van Montfrans (1981) described a rich decapod crustacean community associated with the worm rock habitat. By providing a hard and stable substrate, shelter, and food, worm rock colonies allow many species to inhabit the surf zone that otherwise would be unable to survive there (Gore et al. 1978).

Key Ecological Processes—Worm rock reefs occur within a limited range of depth, water movement, and sediment grain size. Tube-building worms require constant high-energy wave action to supply food, remove waste products, and maintain the suspension of sand grains used for tube building. The worms also require specific grain size sediments to construct the reefs.

Nutrient Sources and Distribution: Tube-building worms are filter feeders, sweeping small suspended organic particles out of the water column. Their food consists primarily of planktonic microorganisms, including diatoms, foraminiferans, and algae.

Detrital Processing and Nutrient Regeneration: The key organisms associated with worm rock reefs consume organic matter carried across them by wave and current action. Their metabolic wastes are in turn carried away by water currents and wave action and become food resources available to other nearshore marine invertebrates.

Habitat Heterogeneity: Worm rock reefs are a mono-specific construction. However, the reefs are irregular in size and shape, providing a heterogenous habitat for the suite of associated organisms. The size and shape of any worm rock reef is dependent on wave energy and the general water movement characteristics of a given location.

Key Natural Disturbances: Wave action, storms, and inundation from beach erosion are the major factors producing natural disturbances in worm rock communities. Wave action is a constant source of disturbance to which the communities have adapted. Storms and significant erosion occur frequently but at irregular intervals.

Landscape Interactions: Worm rock reefs may interact with adjacent sandy beach, subtidal soft bottom, and live bottom communities. Worm rock reefs provide shelter for fishes and motile invertebrates that may feed in adjacent soft bottom areas. Presumably, individual fishes and motile invertebrates may move between worm rock reefs and nearby live bottom areas.

Functional Values—Tube-building worms are the foundation species of a rich and unique nearshore community. Worm rock reefs help stabilize shorelines by diminishing wave energy. They also provide habitat for juvenile life stages of a number of important offshore fish and invertebrate species.

Causes for Deterioration—Worm rock habitat may be altered or destroyed because of sand movement from adjacent areas (e.g., during beach restoration projects), hydrodynamic changes, water quality deterioration, and mechanical damage (e.g., anchoring, boat groundings).

Assessment of Habitat Health—The best indicators of habitat health are the presence, abundance, and health of live tube-building worms. The worm tube disintegrates soon after death of the organism.

Artificial Reefs

Artificial reefs are formed by conglomerations of synthetic or natural material deliberately placed at specific sites in nearshore waters. Artificial reefs may be divided into two categories: 1) those deliberately created as fisheries habitat or for shoreline stabilization and 2) those accidently created, such as shipwrecks or material dump sites. Most artificial reefs have been constructed by fishermen to improve fishing. In 1985, NOAA (1985) published a National Artificial Reef Plan with the objective of "enhancing the habitat and diversity of fisheries resources."

Geographic Distribution—Artificial reefs can be found along any of the coastlines of the United States. Historically, few efforts were made to regulate offshore dumping of debris. Although many states now have active artificial reef programs that require permits specifying precise dumping locations, these are relatively new requirements. The Florida Sea Grant program publishes updated maps of the permitted artificial reefs off Florida and in the eastern Gulf of Mexico. In addition, the California Department of Fish and Game has published an artificial reef plan, with maps showing designated sites off their coastline (Wilson et al. 1990).

Zonation Within Habitats—There is no universal zonation scheme for artificial reefs; zonation primarily depends on the location of the reef. Artificial reefs may be created deliberately or accidentally in water depths ranging from very shallow nearshore areas (shoreline stabilization or shipwrecks) to very deep (deliberately created diving/fishing reefs or shipwrecks). Artificial reefs built specifically to enhance fisheries production have a variety of structural designs; these designs can also influence community zonation (Sheehy 1982). Generally, there is a decreasing biomass of fouling biota with increasing water depth (Gallaway and Lewbel 1982).

Biological Community—Anything that provides vertical relief or structure on an otherwise flat seafloor immediately becomes a focal point for the aggregation of marine species. Once on the bottom, any new substrate is involved in a succession sequence in which filamentous algae attach first, followed by larger algal forms, then sponges and hydroids. If the artificial reef is located in tropical waters, corals will begin to colonize (Ogden and Ebersole 1981). Crustaceans, small fish, and large carnivores also take up residence in the newly available habitat. The artificial reef can provide juveniles of many offshore species with a temporary habitat in which to feed and hide from predation (Doherty and Williams 1988; Bohnsack 1991). In a study of the fish community associated with a nearshore artificial reef off Boca Raton, Florida, Cummings (1990) noted the changes in the particular life stages observed. He attributed these specific life stage absences to changes in food and space requirements, competitive exclusion, and predation. Earlier work by Grant et al. (1982) reported similar life stage changes in fish populations associated with artificial reefs off California.

Key Ecological Processes—Reef structure (including size, vertical relief, and complexity), water depth, and geographic location are among the key factors determining the composition of an artificial reef community. The age of the reef and its proximity to source areas for colonists are also critical because the development of a mature, climax fouling community typically requires several years on newly exposed hard substrates (MRRI 1984).

Nutrient Sources and Distribution: Primary productivity on artificial reefs comes from the algal forms, which begin to attach almost as soon as any hard substrate appears in the marine environment. Nutrient sources include planktonic organisms in the water column and, depending on the location, nutrient runoff from land.

Detrital Processing and Nutrient Regeneration: Filter feeders associated with the artificial reef collect and mineralize plankton and detritus, releasing inorganic nutrients to the algal species present.

Habitat Heterogeneity: There is extensive microhabitat variation within the framework of an artificial reef. Size, shape, and structural heterogeneity of artificial reefs provide multiple areas where small, specialized communities may develop.

Key Natural Disturbances: Depending on their depth, artificial reefs may be affected by coastal erosion, wave action, and storms. Wave action and storms may cause artificial reefs to break apart and scatter across the seafloor. Coastal erosion or sand movement may bury an artificial reef. If artificial reefs are positioned in soft bottom areas, they may eventually be covered over by or sink into these sediments.

Landscape Interactions: Artificial reefs typically occur in soft bottom areas. The new hard substrate interacts with the surrounding environment by providing a place for sessile plants and animals to attach, hiding places for mobile fish and invertebrates, and a concentration of food source organisms for larger predators. Many fishes and motile invertebrates on artificial reefs may feed in adjacent soft bottom areas.

Functional Values—The major functional value of artificial reefs is fisheries enhancement, although in some cases, artificial reefs may be developed to help with shoreline stabilization. With respect to fisheries enhancement, there is some debate as to whether artificial reefs merely attract and aggregate fishes or whether they do, in fact, produce additional fish biomass (Bohnsack and Sutherland 1985; Bohnsack 1989). There are at least five ways that offshore structures can enhance fish biomass (Bohnsack 1989): 1) provide additional food, 2) increase feeding efficiency, 3) provide shelter from predation, 4) provide recruitment habitat for settling individuals that would otherwise have been lost to the population, and 5) vacate space in the natural habitat, thus allowing replacement from outside the system.

Causes for Deterioration—Artificial reefs may deteriorate as a result of waves and currents, particularly if the structure is a loose aggregation (e.g., tire reefs). Artificial reefs may also be damaged by trawling, dredging, or anchoring. In soft sediments, artificial reefs may be buried or sink into the seafloor.

Assessment of Health Habitat—Potential indicators of habitat health in artificial reefs include the size and relief of exposed structures; abundance of attached epibiota; and abundance of associated fish populations.

Algal Communities

Geographic Distribution—Nearshore subtidal macroalgal communities occur along the entire coastline of the continental United States Throughout a large part of the cold temperate coastal region, algal communities are dominated by large brown algal species known collectively as kelps. *Laminaria* sp. is the dominant kelp species in Atlantic waters, while Pacific waters are dominated by *Macrocystis* sp. (Nybakken 1988). Off the coastline of Florida, algal communities are formed by the calcareous green algal species *Halimeda* sp., *Penicillus* sp., *Rhipocephalus* sp., and *Udotea* sp. (Multer 1969).

Zonation Within Habitats—Horizontal zonation patterns of algal communities are similar along all coasts. Short, mat-forming species grow in the wave zone, followed by larger, frond forming species further offshore. Eventually, as depth increases, smaller filamentous and encrusting species again become the dominant forms (Foster and Schiel 1985; Nybakken 1988; CSA 1989). Figure 5A-4 illustrates the typical vegetative layering zones in Pacific coast kelp forests.

Biological Community—Macroalgal community structure varies considerably by geographic region. In the north Atlantic region, the subtidal macroalgal community is extensive and is generally characterized by large perennial kelp species such as *Laminaria* sp. and *Agarum* sp. In the mid-Atlantic area, the macroalgal community is reduced in terms of its overall distribution and biomass, primarily because of the lack of colonizing substrate (hard bottom) along this coast. In the warm waters of south Florida and the Gulf of Mexico, the macroalgal community is diverse and widespread, but no single group of species defines the habitat. The community does not reach the biomass levels that occur in the north-Atlantic or Pacific, primarily because of the lack of a rocky coast. In waters off southern California, the macroalgal community, dominated by the kelp species *Macrocystis pyrifera*, is not as diverse or as abundant as that off central and northern California. Kelp forests off central and northern California

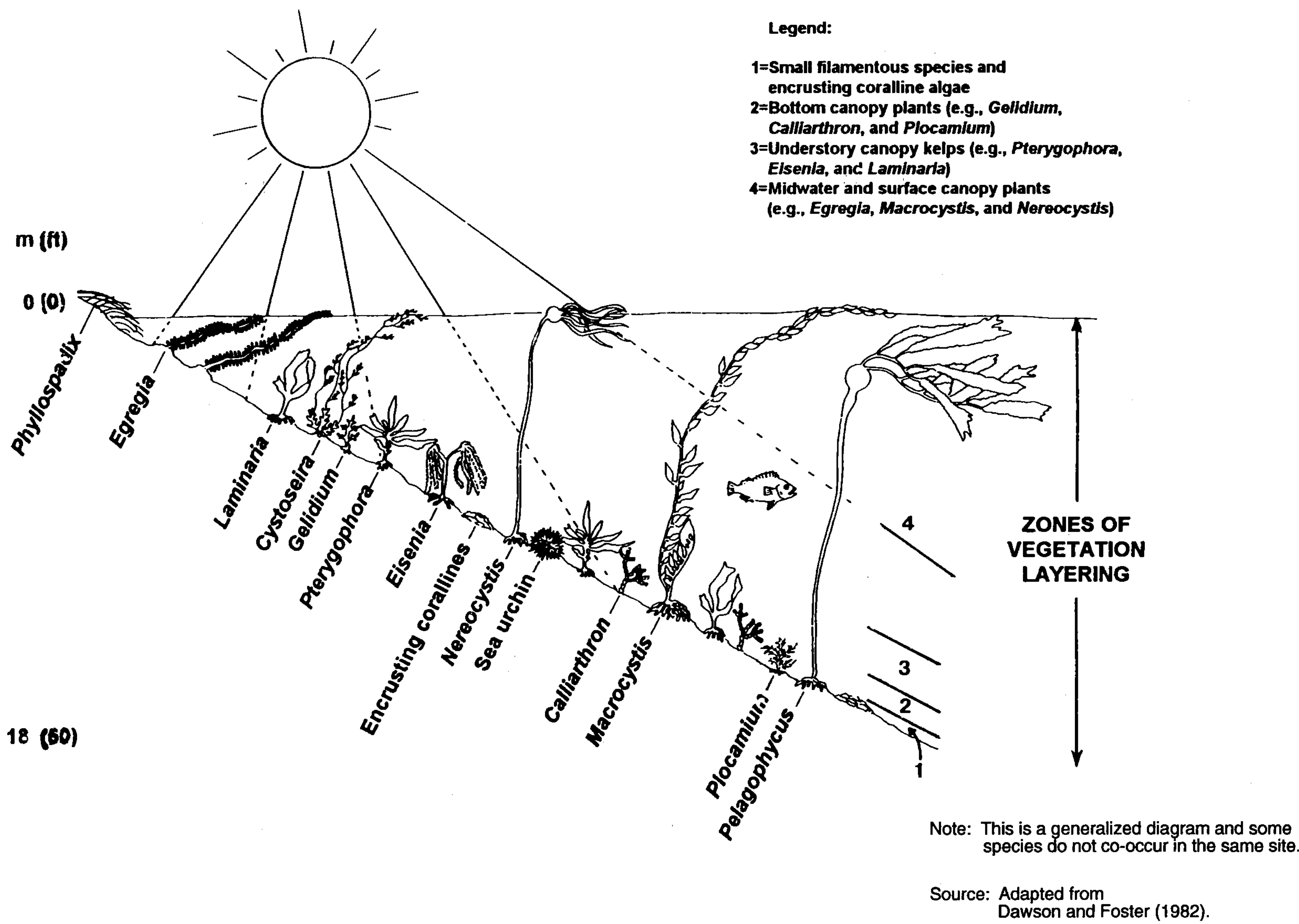
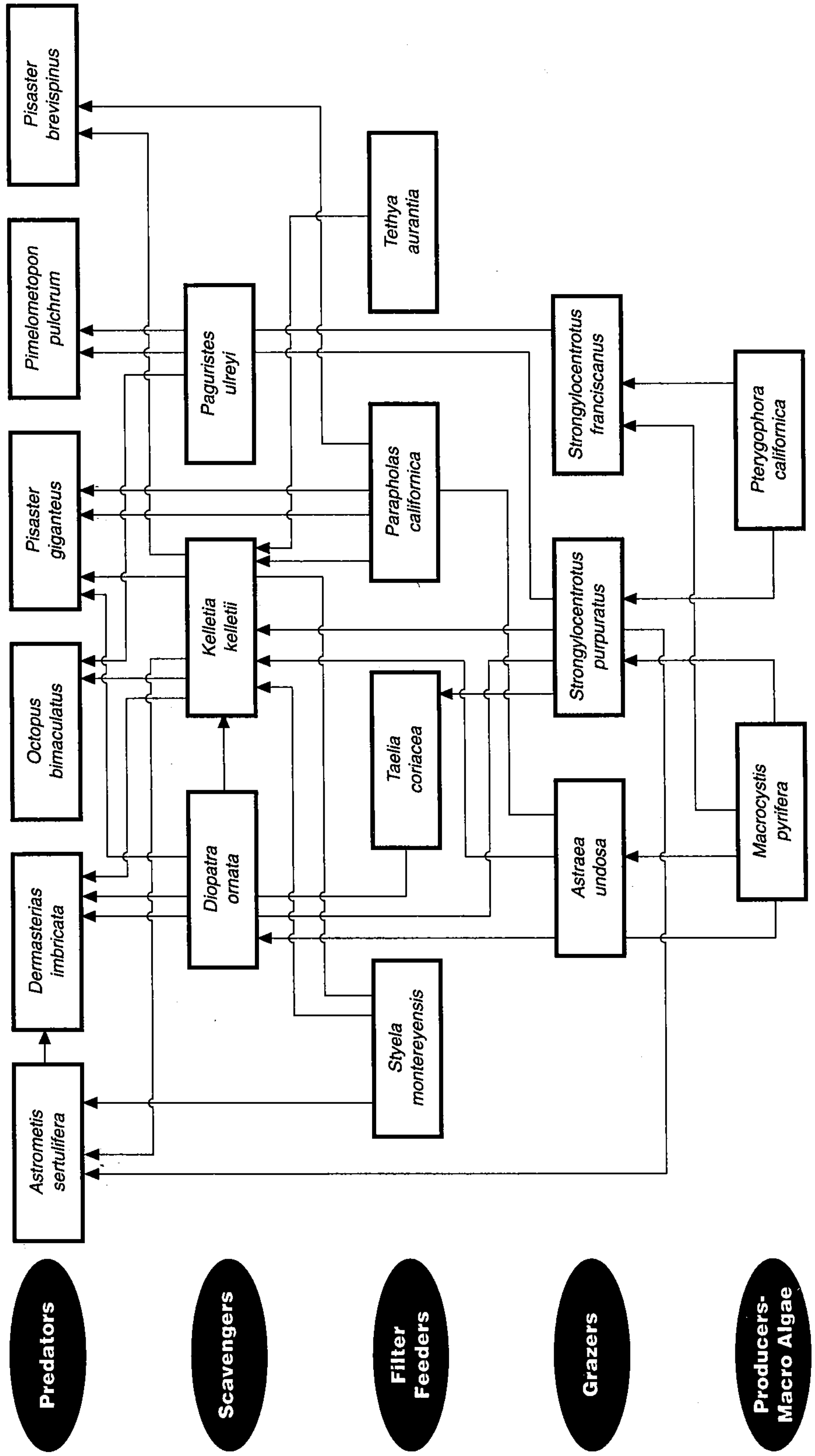


FIGURE 5A-4. Typical vegetation layering zones of a Pacific coast kelp forest.

are extremely productive because of the California current and cool, nutrient-rich, upwelling conditions. Much of this coastline is rocky and thus provides an excellent substrate for macroalgal growth. A rich macroalgal community dominated by kelp species also exists in waters off the coasts of Washington and Oregon. Most of the research on kelp species in this area has been directed toward the commercially important species *Iridaea cordata* and *Gigartina exasperata*. *Laminaria* sp. dominates the Alaskan macroalgal community primarily because of their ability to continue growing during long periods of darkness (Foster and Schiel 1985).

Giant kelp forests are inhabited by an abundant and species-rich invertebrate fauna. In giant kelp holdfasts alone, Andrews (1945) found more than 23,000 individuals representing nine phyla from collections in Monterey, California. Foster and Schiel (1985) organized groups of invertebrates from kelp forests functionally by feeding group in an effort to obtain ecological relevance; however, some of the organisms in question overlapped groups. Invertebrate suspension and detritus feeders include sponges, which along with tunicates and bryozoans, are one of the most common sessile animals in kelp forest; cnidarians (hydroids, sea anemones, solitary polyp corals, hydrocorals, and gorgonians); bryozoans; echinoderms (brittle stars, sea stars, sea cucumbers, and sea urchins); molluscs; polychaete worms; sipunculids (peanut worms); crustaceans including isopods (*Idotea* sp.), hermit crabs (*Pagurus* sp.), spider crabs (*Cancer* sp.), and barnacles (*Balanus* sp.); and tunicates (primarily *Styela* sp.). Grazers include sea urchins (*Strongylocentrotus franciscanus*, *Sebastes purpuratus*, and *Lytechinus anamesus*) and sea stars (*Patiria miniata*); gastropods (*Tegula* sp., *Calliostoma* sp., *Mitrella* sp., *Lacuna* sp., *Norrisi* sp. and *Astraea* sp.); and crustaceans, including species of amphipods (*Taliepus nutalli*) and crabs (*Pugettia producta*). Fish species associated with kelp forest include senorita (*Oxyjulis californica*), kelp surfperch (*Brachyistius frenatus*), halfmoon (*Medialuna californiensis*), garibaldi (*Hypsypops rubicundus*), California sheepshead (*Semicossyphus pulcher*), opaleye (*Girella nigricans*), blue rockfish (*Sebastes mystinus*), blacksmith (*Chromis punctipinnis*), giant kelp fish (*Heterostichus rostratus*), kelp bass (*Paralabrax clathratus*), olive rockfish (*Sebastes serranoides*), black rockfish (*Sebastes melanops*), greenlings (*Hexogrammos* sp.), lingcod (*Ophiodon elongatus*), and cabezon (*Scorpaenichthys marmoratus*). Figure 5A-5 shows a simplified food web from a southern California kelp forest (Foster and Schiel 1985).

Key Ecological Processes—The universal limiting factors for hard bottom macroalgal community distribution throughout the coastal waters of the United States include light availability, nutrient availability, water clarity, temperature, exposure to wave action, and the presence of hard substrate for attachment. It



Source: Modified from Rosenthal et al. (1974).

FIGURE 5A-5. Food web of a southern California kelp bed.

is difficult to determine which is the key factor in any specific community, but it is easy to demonstrate the relationship between any one factor and the distribution of algal species.

Nutrient Sources and Distribution: Macroalgal species must obtain all their nutrients from the seawater around them. Their holdfasts are purely for attachment to the substrate and serve no purpose in nutrient uptake. Most macroalgal species are never completely exposed to air and all metabolic process must occur in the water. Nutrient sources are the upwelling of cool, nutrient rich water along the continental shelf break and terrestrial runoff.

Detrital Processing and Nutrient Regeneration: Detritus is processed by the many filter feeding and grazing invertebrates that live in association with macroalgal communities. Some of the inorganic nutrients they release into the water column may be used by the macroalgal species present.

Habitat Heterogeneity: Kelps or other large macroalgal species form the basic structure of the macroalgal community. These species generally form a "canopy," and below them is another distinct set of "understory" algal species. Habitat heterogeneity within the macroalgal community depends on the density of "canopy" vs. "understory" algal species present. Individual algal plants have a number of microhabitats within their fronds and holdfast systems.

Key Natural Disturbances: Storms that displace algal holdfasts and predation by invertebrates are the major natural disturbances within macroalgal communities. Storms occur infrequently but with regularity along all coastlines. Normal population dynamics maintain a balance between the community-forming macroalgal species and predators who consume young plants. Macroalgal communities are damaged or destroyed when one or more of their natural predators undergoes a population explosion. Once a macroalgal community has been destroyed, it is often difficult to reestablish.

Landscape Interactions: Macroalgal communities interact with the open ocean waters around them, with adjacent soft bottom and hard bottom communities, and with the terrestrial shoreline.

Functional Values—Macroalgal communities provide organic material (detritus) that is distributed across the entire shallow shelf. They also provide temporary shelter and food for many commercially valuable juvenile open ocean fish and invertebrates. They absorb wave energy and help stabilize the shoreline. Some species of kelp are commercially harvested.

Causes for Deterioration—Algal communities may be altered or destroyed by deteriorating water quality or damage from waves and storms.

Assessment of Habitat Health—Potential indicators of habitat health in macroalgal communities include water clarity; nutrient concentrations and ratios in water, sediment, and tissue; epiphyte loads; biomass of macroalgal foundation species; and contaminant concentrations in sediment and tissue.

Seagrass Beds

Geographic Distribution—Seagrass beds are found in sheltered areas on both the East and West coasts of the United States and in the Gulf of Mexico, but their extent and species composition varies considerably. Eelgrass (*Zostera marina*) communities occur from the United States/Canadian border to the Virginia/North Carolina border, but they are primarily restricted to inshore waters, such as behind the New Jersey barrier islands and in Chesapeake Bay.

Along the Southeastern coast from Cape Hatteras throughout the Gulf of Mexico, seagrasses are mostly confined to estuaries and lagoons behind barrier islands. However, extensive seagrass beds occur in open near coastal waters along the Florida Keys, in Florida Bay, and in the Florida Big Bend area (Zieman 1982; Zieman and Zieman 1989). Florida and Gulf of Mexico seagrass beds are dominated by turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), or shoal grass (*Halodule wrightii*), depending on the location (CSA and MLI 1986; CSA 1989).

On the West coast, seagrass beds occur in sheltered locations from Vancouver Island to Baja, California. There are five species of seagrass found along the Pacific coast. Three are surfgrasses (*Phyllospadix scouleri*, *Phyllospadix torreyi*, and *Phyllospadix serrulatus*), and the other two are eelgrass (*Zostera marina* and *Zostera japonica*). *Phyllospadix scouleri* and *Phyllospadix torreyi* range from the northern end of Vancouver Island to the southern end of the Baja California. *Phyllospadix serrulatus* extends northward from Oregon around the southern coastline of Alaska (Phillips 1984).

Zonation Within Habitats—Shoal grass is so named because it can grow in shallower water than most of the larger seagrass species. Along the North-eastern coast, shoal grass is typically found closest to shore, and *Z. marina* is found in slightly deeper water. Along the southeast Atlantic coast and Gulf of Mexico, shoal grass is typically found in the shallowest water, followed by turtle grass or manatee grass, depending on the location, with an outer fringe of shoal grass. On the Pacific coast, *Phyllospadix* spp. grows closest to shore, along with *Zostera japonica*. *Zostera marina* is found in deeper water (Phillips 1984; Thayer et al. 1984; CSA and MLI 1986; CSA 1989).

Biological Community—Seagrass communities, whether temperate or subtropical, usually have one or two dominant seagrass species. They are complex communities consisting of large numbers of epiphytic and epizoic organisms, burrowers, invertebrates, and fish. Individual species making up a seagrass community vary with geographic location, particularly latitude.

Along the Northeastern and mid-Atlantic coasts of the United States, there are three distinct community types of benthic and epibenthic fauna associated with eelgrass beds. Cape Cod and Cape Hatteras are the generally accepted biographic boundaries for these communities, and while many of the individual species change, many genera remain constant. These constant species include gastropods (*Bittium* sp. and *Anachis* sp.), isopods (*Erichsonella* sp.), and shrimp (*Hippolyte* sp.). Major infaunal species include lamellibranch bivalves (*Tellina* sp.), polychaetes (*Nereis* sp.), and amphipods (*Corophium* sp.) (Thayer et al. 1984).

Eelgrass beds in the mid-Atlantic region provide nursery habitat for decapod crustaceans and fishes, including the commercially important blue crab (*Callinectes sapidus*) and summer flounder (*Paralichthys dentatus*). Resident fishes of eelgrass beds include the northern pipefish (*Syngnathus fuscus*), the darter goby (*Gobionellus boleosoma*), and the feather blenny (*Hypsoblennius hentzi*). Seasonal residents include juvenile spot (*Leiostomus xanthurus*), silver perch (*Bairdiella chrysoura*), and pinfish (*Lagodon rhomboides*). Large, transient predators associated with Atlantic coast eelgrass communities include the cownose ray (*Rhinoptera bonasus*), sandbar shark (*Carcharinus plumbeus*), bluefish (*Pomatomus saltatrix*), weakfish (*Cynoscion regalis*), and spotted seatrout (*Cynoscion nebulosus*). Considerable diel and seasonal variation in abundance is characteristic of fish and decapod communities in eelgrass beds (Thayer et al. 1984)

Along the southeast Atlantic coast and throughout the Gulf of Mexico, seagrass community structure is determined by the species composition and density of seagrasses present, as well as a number of abiotic factors. There are three species of perennial, bed-forming seagrasses offshore in south Florida and the Gulf of Mexico: turtle grass, shoal grass, and manatee grass. These species form the vast seagrass meadows along the open coastlines of Florida Bay and the Florida Big Bend. There are also two species of annual seagrasses, *Halophila engelmanni* and *Halophila decipiens*, that are seen seasonally across large areas of the west Florida continental shelf in deeper water (10–36 m [30–120 ft]), but these species do not form persistent beds. The most obvious invertebrates associated with south Florida seagrass beds include the queen conch (*Strombus gigas*), the Bahamian starfish (*Oreaster reticulata*), gastropods (*Fasciolaria tulipa* and *Pleuroploca gigantea*), and numerous species of sea urchins, including *Lytechinus variegatus* and *Tripneustes ventricosus*. Juveniles of the long-spined sea urchin (*Diadema antillarum*) are also seen in seagrass beds, but the adults are more common on the rock ledges of the offshore coral reefs. Other prominent members of the southern seagrass invertebrate community include holothurians (*Actinopyga agassizi* and *Holothuria floridana*), the nudibranch (*Aplysia dactylomela*), the pink shrimp (*Penaeus duorarum*), the Florida lobster (*Panulirus argus*), and the octopus (*Octopus briareus*) (Zieman 1982).

Resident fishes in southern seagrass beds include pipefish (*Syngnathus* spp.), seahorses (*Hippocampus zosterae* and *Hippocampus erectus*), and the inshore lizardfish (*Synodus foetens*). Gobies and clinids, especially the code goby (*Gobiosoma robustus*) and the banded blenny (*Paraclinus fasciatus*) are abundant. Ophichthid eels (*Myrichthys* spp.) are common daytime foragers in the grass beds, while the blackedge moray (*Gymnothorax nigromarginatus*) is active primarily at night. Characteristic seasonal grassbed residents in northeast Gulf waters include spot and silver perch. Grunts (Haemulidae) are well-represented in south Florida waters, as are snappers (Lutjanidae) and mojarras (Gerreidae). Pinfish display a strong affinity for subtropical seagrass beds and are often the numerically dominant species present. Spotted seatrout are abundant predators in and around southern seagrass beds and are often seen chasing large schools of striped mullet (*Mugil cephalus*) or other forage species (Zieman 1982; Zieman and Zieman 1989).

Eelgrass communities along the Pacific Northwest coast have similar components, including scallops, crabs, sponges, mussels, sea urchins, starfish, flatworms, polychaete worms, nematodes, and amphipods, as those found in East coast eelgrass communities. Individual species vary considerably, but most major phyla are represented in similar proportions. In California and along the Pacific Northwest coast through Alaska, many commercially important species, such as dungeness crabs (*Cancer magister*), coon-striped shrimp (*Pandalus dance*), English sole (*Parophrys vetulus*), and starry flounders (*Platichythys stellatus*), are associated with eelgrass meadows. There are no definitive publications that describe specific patterns of eelgrass community variation with latitude along the Pacific coast (Phillips 1984).

Key Ecological Processes—Seagrass beds are strongly influenced by several physical and chemical factors. Water depth and clarity control the depth distribution and species zonation of seagrass beds. Seagrass beds typically occur in oligotrophic waters because high nutrient concentrations favor the growth of phytoplankton (which reduce light reaching the bottom) and seagrass epiphytes. Temperature controls their latitudinal distribution, as well as their growth and respiration rate, and salinity controls their species composition with respect to distances from freshwater inputs.

Nutrient Sources and Distribution: As rooted plants, seagrass species absorb nutrients from the sediments or substrate. Therefore, they are capable of recycling nutrients into the ecosystem that would otherwise be unavailable.

Detrital Processing and Nutrient Regeneration: Little of the energy reserve produced through seagrass photosynthesis enters the food chain directly because few species graze on seagrasses. Instead, seagrass energy reserves enter the system as detritus after the plant dies. These detrital particles are carried across continental shelves and enrich areas that are considerable distance from the seagrass beds.

Habitat Heterogeneity: Seagrass beds are much more heterogenous than adjacent soft bottom areas and provide numerous microhabitats for specialized organisms. Habitat heterogeneity within seagrass beds depends on the density of the seagrass shoots in a given area. There is also a "canopy/understory" effect in seagrass beds formed by the long-bladed species.

Key Natural Disturbances: Storms and disease are two major factors producing natural disturbances in seagrass beds. Storms occur at intervals along all North American coastlines. Massive diseases, or epidemics, in seagrass communities are rare, but when they occur they can have catastrophic consequences for seagrass beds. In the early 1930s, a "wasting disease," caused by the slime mold *Labyrinthula macrocystis* destroyed between 80 and 90 percent of the *Zostera marina* beds in the North Atlantic. The impacts from this environmental disaster are still evident in many northeastern estuaries.

Landscape Interactions: As primary producers, seagrass beds interact with benthic communities across the entire continental shelf. Seagrass beds provide nursery habitat for fish and invertebrates that are seen and harvested in all nearshore waters. Seagrass beds interact with their adjacent shoreline and shoreline communities by stabilizing bottom sediments, recycling nutrients, and improving water quality.

Functional Values—Seagrass beds are extremely important nursery areas, providing shelter and food to larvae and juveniles of commercially and recreationally valuable fish and invertebrate species. Seagrass beds help stabilize bottom sediments, thereby improving water quality and helping to prevent coastal erosion. Seagrass beds are also valuable because they produce organic matter (plant tissue) that may be exported as detritus to adjacent systems.

Causes for Deterioration—Historically, the greatest losses of seagrass beds have resulted from dredge-and-fill projects in coastal waters. Today, the main cause of further habitat loss is poor water quality in the form of turbidity and increased nutrient concentrations (which promote phytoplankton and epiphytic growth).

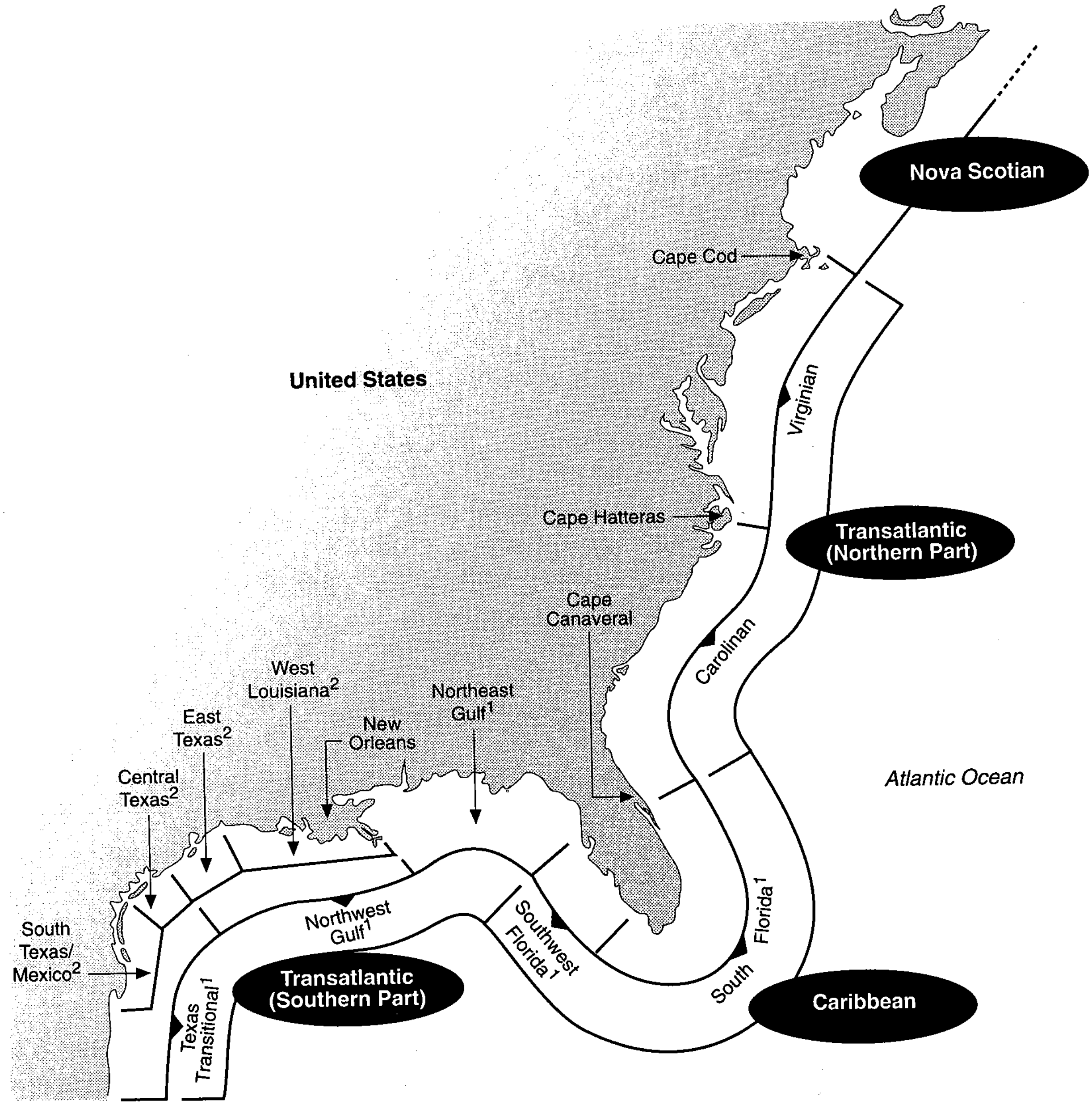
Assessment of Habitat Health—Potential indicators of habitat health in seagrass beds include water clarity; nutrient concentrations and ratios in water, sediment, and tissue; epiphyte loads; short shoot densities; and contaminant concentrations in sediment and tissue.

Non-Vegetated Soft Bottom Communities

Geographic Distribution—Non-vegetated subtidal soft bottom communities are the most common bottom communities around the entire coastline of the United States. Figure 5A-6 shows the general biotic provinces along the eastern seaboard and Gulf of Mexico coastlines of the United States, and Figure 5A-7 shows the general biotic provinces of the Pacific coast. Non-vegetated soft bottom communities fall into these general subtidal community types.

Zonation Within Habitats—Non-vegetated subtidal soft bottom communities occur from the intertidal to abyssal depths (Coull 1977). Zones corresponding with depth and distance from shore are often recognized (Thorson 1957; Rabalais and Boesch 1987). Shallow subtidal soft bottom habitats often show an increasing species richness with distance from shore. Depth and distance from shore are usually surrogates for factors such as sediment texture, temperature (and range), salinity (and range), light, and productivity of the overlying water column.

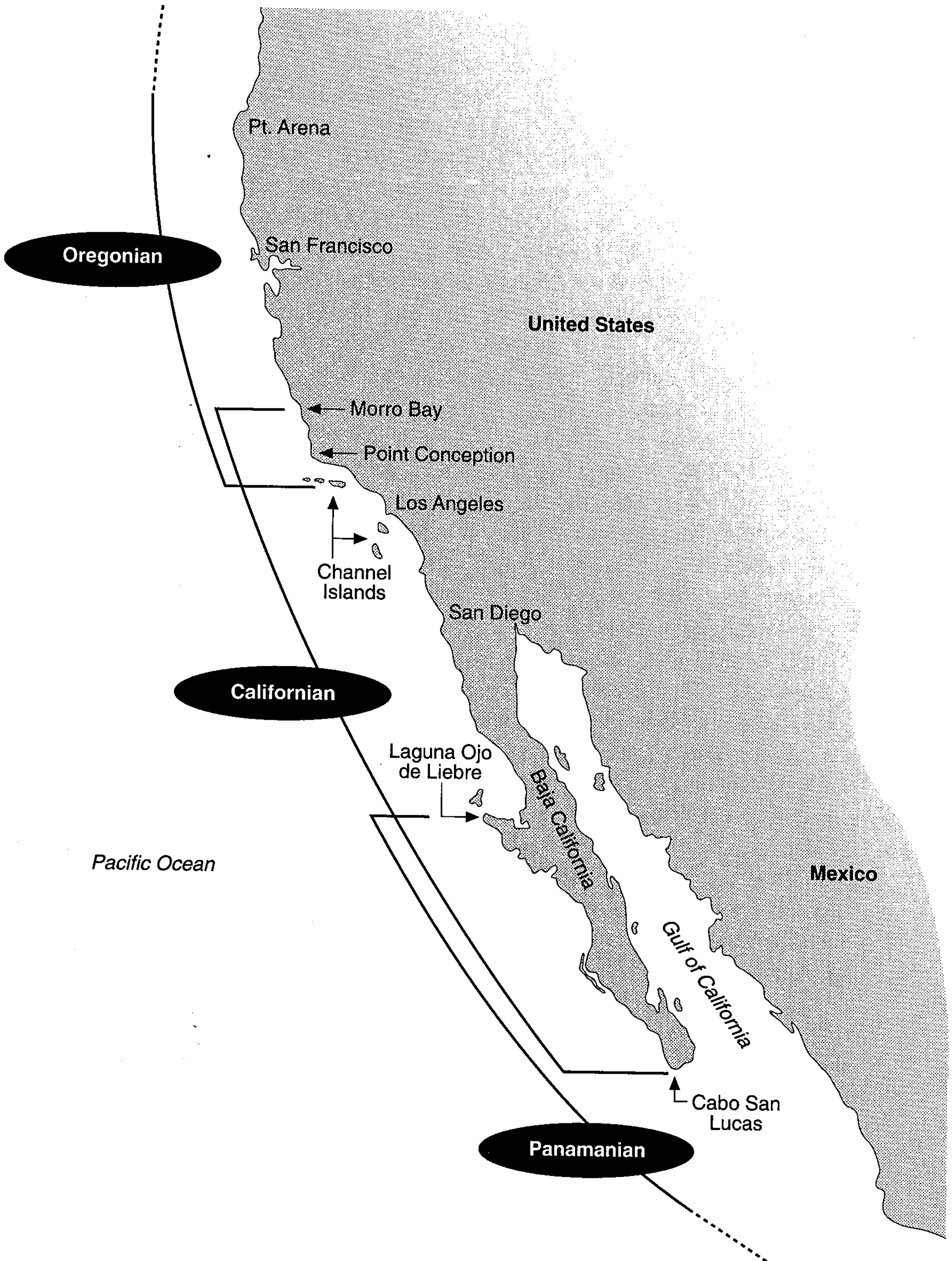
Biological Community—Non-vegetated soft bottom communities include both epifaunal and infaunal organisms. Epifauna are organisms living on the substrate, whereas infauna live within the sediments.



Source: Modified from Valentine (1963).

[1] Provinces described by Pulley (1952)
[2] Provinces described by Parker (1960)

FIGURE 5A-6. Biotic provinces proposed by various biogeographers for the Atlantic and Gulf coasts of the United States.



Source: Modified from Valentine (1963).

FIGURE 5A-7. Biotic provinces in general use for the Pacific coast of the United States and northern Mexico.

Along the Atlantic coast, these communities are dominated by a relatively even mix of polychaetes, bivalves, gastropods, and amphipods (Rabalais and Boesch 1987). A non-vegetated community off the New England coast at 60 m (196 ft) includes polychaetes (*Polygordius* sp.), lamellibranch bivalves (*Tellina* sp.), and amphipods *(Pseudunciola* sp. and *Protohaustorius* sp.) as the most common species present. Along the mid-Atlantic coast, the dynamic sandy bottoms of the inner shelf are numerically dominated by small interstitial feeders such as tanaids (*Tanaissus* sp.), and polychaetes *(Polygordius* sp., *Goniadella* sp., and *Lumbrinerides* sp.). Along the south Atlantic coast, communities are dominated by polychaetes (primarily spionids), gastropods (*Oliva* sp., *Terebra* sp.), portunid crabs (*Arenaeus* sp., *Callinectes* sp., and *Ovalipes* sp.), and burrowing shrimp (*Callianassa* sp.). In slightly deeper water, the fauna is dominated by polychaetes, haustorid and other amphipod groups, and bivalves such as *Donax* sp. and *Tellina* sp.

The Florida Keys are in a transitional zone between the physiographic provinces of the Atlantic Ocean and the Gulf of Mexico. Shallow water (<40 m; 131 ft) soft bottom communities on the west Florida shelf are dominated by polychaetes (*Vermiliopsis* sp., *Fabricia* sp., *Hydroides* sp., *Lumbrineris* sp., *Goniadides* sp., *Pisione* sp., *Ehlersia* sp. and *Prionospio* sp.), gammarid amphipods (*Maera* sp. and *Photis* sp.), cumaceans *Cyclaspis* sp.), and caprellids (*Phtisica* sp.) The north central Gulf of Mexico from Cape San Blas to the Mississippi Delta is characterized by low species diversity. This is attributed to the higher sedimentation rates and finer grain size sediments seen in that area. The dominant species reported from that area are syllid polychaetes (*Syllis* sp. and *Sphaerosyllis* sp.), nereid polychaetes (*Websterinereis* sp.), segmented worm (*Glycera* sp.), *Lumbrineris* sp., *Paraprionospio* sp., *Prionospio* sp., and capetellid polychaete worm (*Mediomastus* sp.). The northwestern Gulf of Mexico (Mississippi Delta to Galveston, Texas) is characterized by the large freshwater discharges of the Mississippi and Atchafalaya Rivers. The nearshore soft bottom community there is dominated by polychaetes, crustaceans, and bivalves (Rabalais and Boesch 1987).

The topography and sediments off southern California provide a complex array of benthic habitats. In the shallow nearshore zone ≤25 m (82 ft) with coarse-grained sediments, the brittle star *Amphipholis* sp. is the most abundant organism present. This community is also described as a *Nothria-Tellina* association in which *Diopatra ornata* and *Prionospio* sp. are also conspicuous elements. Other common taxa are gastropods (*Olivella* sp.), cumaceans (*Diastylopsis* sp.), and amphipods (*Paraphoxus* sp.). On finer sediments in deeper water, another brittle star, *Amphiodia urtica*, was the numerically dominant species. Here echiurans (*Listriolobus* sp.), brachiopods (*Glottida* sp.), pelecypods (*Axinopsida* sp., *Mysella* sp., and *Parvilucina* sp.), ostracods *(Euphilomedes* sp.) and a variety of gammarid amphipods are the most common groups seen (Rabalais and Boesch 1987).

The subtidal habitats seen from Point Conception to the California/Oregon border are communities of the tube worm *Diopatra ornata* in shallow, coarse sandy habitats and communities of another tube worm, *Nothria elegans*, in finer sands. The sand dollar (*Dendraster excentricus*), is also characteristic of a variety of sand habitats (Rabalais and Boesch 1987).

Off the Washington/Oregon coast, there is a shallow water (<36 m; 118 ft) sand bottom community dominated numerically by cumaceans (*Diastylopsis* sp.). Amphipods (*Ampelisca* sp. and *Paraphoxus* sp.), lamellibranch bivalves (*Tellina* sp. and *Macoma* sp.), and polychaete (*Owenia* sp.) are also abundant. In deeper waters, the most abundant species were polychaetes (*Sternaspis* sp., *Magelona* sp., *Nephtys* sp., and *Haploscoloplos* sp.), and lamellibranch bivalves (*Yoldia* sp. and *Axinopsida* sp.) (Rabalais and Boesch 1987).

Key Ecological Processes—Species composition within non-vegetated soft bottom communities is primarily dependent on sediment characteristics such as grain size, sorting, porosity, particle shape, and packing (Gray 1981). Wave action and currents play a critical role in sorting sediments with respect to grain size. Salinity, temperature, and light penetration through the water column are also critical environmental factors for nearshore soft bottom communities, but organisms living in these environments are generally adapted for rapid fluctuation in these parameters. Dissolved oxygen can be a critical factor in shallow benthic systems. Near bottom concentrations can be depleted in summer months, leading to anoxia and death of the macrofaunal community.

Nutrient Sources and Distribution: Upwelling, terrestrial runoff, and primary productivity within adjacent seagrass and algal communities provide nutrients to the non-vegetated soft bottom areas. Because nearshore waters are well mixed, nutrients are rarely limiting or bound up in bottom reservoirs.

Detrital Processing and Nutrient Regeneration: A large amount of detrital processing by suspension feeders and deposit feeders takes place in soft bottom communities. Depending on community location, detritus originates from adjacent benthic algal or seagrass areas, phytoplankton in the water column, and/or terrestrial runoff.

Habitat Heterogeneity: By comparison with rocky habitats and seagrass communities, non-vegetated soft bottom areas show little heterogeneity on the surface. However, there is significant small-scale heterogeneity, including microhabitats resulting from burrows, mounds, and biological disturbance. Because sediment size and physical geomorphology play such an important role in community development, soft bottom communities tend to be complex mosaics of slightly different species assemblages. Variability among benthic samples is always high, and in some areas, such as the northeastern Gulf of Mexico and southern California, community variation caused by substrate morphology is significantly greater than community variation that is the result of latitude or depth.

Key Natural Disturbances: Storms, turbidity currents, and earthquake-caused sediment slumping all cause natural disturbances within the soft bottom community. These types of disturbances occur frequently but at irregular intervals.

Landscape Interactions: Non-vegetated soft bottom habitats are the dominant feature of the subtidal landscape. They are linked by flows of energy, material, and individuals with adjacent vegetated communities, rocky and sandy beach communities, and the nektonic communities of the water column. Most of the detritus from the water column and intertidal habitats eventually settles to soft bottom, depositional environments. Many fishes and invertebrates in hard bottom communities feed on adjacent soft bottom infauna.

Functional Values—Many commercially valuable species of fish and invertebrates rely on benthic infauna and epifauna for food. In fact, modern benthic ecology began as the study of "the sea bottom and its production of fish food" (Petersen 1918). Much of the nutrient recycling and secondary productivity on continental shelves takes place within the soft bottom community.

Causes for Deterioration—Subtidal soft bottom communities may be destroyed or damaged by dredging, burial with dredged material from other locations, and/or water quality deterioration.

Assessment of Habitat Health—Potential indicators of habitat health in non-vegetated, subtidal soft bottom communities include infaunal abundance, diversity, and species composition (especially abundance of pollution indicator species); concentrations of contaminants in sediment; and depth of redox layer. Sediment profile imaging provides a useful tool to identify and characterize the status of disturbed soft bottom communities.

KEY ENVIRONMENTAL PARAMETERS

Table 5A-3 summarizes key environmental parameters that should be considered in implementing restoration projects in open coastline and near coastal waters. In all of the open coastline and near coastal water habitats, substrate characteristics are critical to the success of restoration projects. The projects should attempt to replace or mimic the natural substrate as closely as possible. Many systems also depend on particular tidal and flow regimes that are affected by the placement and orientation of the substrate.

Ecosystems that are particularly sensitive to water quality conditions are coral reefs and live bottom, algal communities, and seagrass beds. Where water quality problems such as turbidity, nutrients, or toxic contaminants have caused or contributed to habitat loss, management actions are needed to improve water quality or restoration/rehabilitation projects will ultimately fail.

RESTORATION PROJECTS

Potential restoration projects along open coastlines and in near coastal waters are summarized in Table 5A-4. The first column lists problems that may cause habitat loss, damage, or deterioration in each type of habitat. For each problem, one or more possible solutions is given in the second column.

Most of the activities listed in Table 5A-4 could be considered as either restoration or rehabilitation, depending on the scope of a specific project. For example, pumping sand onto a beach could be considered *rehabilitation* if the project simply made the beach "look normal" and serve human uses. In contrast, a *restoration* project would seek to ensure that the sand was similar in composition and that the fauna and ecological processes of the restored beach were similar to those observed prior to disturbance. Activities to minimize or avoid further beach erosion would be considered *management*.

Widespread causes of damage and deterioration in open coastline and near coastal habitats include dredging, dredged material disposal, and deteriorating water quality (particularly in terms of nutrients, reduced water clarity, and increased concentrations of toxic contaminants). Erosion, a natural process that can be exacerbated by human structures in the coastal zone (e.g., inlets, jetties), causes significant loss of beach habitat in some areas. Environmental management considerations in conducting beach nourishment projects should be directed toward ensuring that other important marine communities or coastal habitats are not damaged while restoring the beach. During any dune restoration or rehabilitation project, care must be exercised not to damage or destroy critical habitat for endangered and threatened species and species of special concern. Trapping and

TABLE 5A-3. KEY ENVIRONMENTAL PARAMETERS ALONG OPEN COASTLINE AND NEAR COASTAL WATERS

Parameter	Comment
Beach and Intertidal Habitats	
Rocky Shorelines	
Water quality	Organisms that live on rocky shores are dependent on seawater to provide nutrients and remove waste products. Changes in water quality along a rocky coast may be reflected in changes in the types of organisms making up the rocky shore community.
Tidal patterns and wave action	Tidal highs and lows and wave action control the zonation patterns on rocky shores. Alterations in tidal flow patterns or shoreline profile will alter the widths of the community zones.
Substrate type	Rock type and size play an important role in defining a rocky shoreline community. If a rocky intertidal community is to be restored or rehabilitated, the rehabilitated substrate should match the original community as nearly as possible.
Sandy Beaches and Sand Dunes	
Beach front profile	Wave action and the beach front profile, or angle of a beach foreshore with respect to the water, are critical factors in defining the environment of a beach. Restored beach profiles should match the profiles of the natural beach at that location as nearly as possible.
Particle size and compaction parameters	Particle size and compaction parameters are vitally important to the many species of organisms that burrow into a beach or dune area. They are also critical considerations in terms of marine sea turtle nesting and the ability of turtle hatchlings to escape their nest once out of the shell.
Dune stabilization and revegetation	If mechanically damaged or eroded sand dunes are to be restored, they must be revegetated immediately to help prevent further erosion. Reconstructed dunes must be stabilized in some manner while replanted vegetation is given time to develop.
Subtidal Habitats	
Coral Reefs and Live Bottom Areas	
Water quality	Coral reef and live bottom community species are extremely sensitive to water clarity, nutrients, temperature, and salinity. Changes in these parameters may change species composition of the community at a given location.

TABLE 5A-3. (cont.)

Parameter	Comment
Subtidal Habitats (cont.)	
Substrate type	Coral reef and live bottom community species require a hard substrate for attachment and growth. Most species in this community attach to a limestone base substrate. Any planned restoration or rehabilitation of these communities by the construction of artificial hard bottom habitats should attempt to use natural substrate material if possible.
Habitat location and stabilization	Proper placement and stabilization of artificially created hard bottom habitat is important. New hard bottom must be located in an area where it will not sink into the sediments, and it must be stabilized to a point where it will not be moved about by wave surge and storms.
Worm Rock Reefs	
Water flow patterns	Worm rock reefs are dependent on fast moving water and a steady supply of suspended sand in order to develop. They must be located in the surf zone or in fast moving tidal inlets.
Substrate particle size	Sabellariid worms that build worm rock reefs are restricted in the type and size of the sand grains they can cement together. If the particle size and composition of the substrate is changed, the community cannot be reestablished.
Transplanting stock	Worm rock can be transplanted from one location to another to help restore a worm rock reef community. Other environmental conditions must be right for such a community to be viable.
Initial stabilization	If transplantation is attempted or if hard substrate is placed in the surf zone to facilitate worm rock recolonization, the transplanted worm rock or hard substrate fragments must be mechanically anchored or stabilized to prevent shifting and fragmentation in the surf zone.
Artificial Reefs	
Substrate type	If an artificial reef community has been completely destroyed and restoration is desirable, then the new artificial reef material should match the original material as closely as possible.
Location and stabilization	To be viable for any length of time, all artificial reefs must be located in areas where there is a hard substrate near enough to the surface to prevent the artificial reef material from sinking completely into the sediments. Artificial reef material placed in near coastal waters needs to be mechanically anchored or stabilized to prevent shifting by wave action or storm surges.
Water quality	Nearshore algal communities are sensitive to water quality parameters (water clarity, nutrients, temperature, and salinity). Long-term modifications to water quality will change the species composition of the community.

TABLE 5A-3. (cont.)

Parameter	Comment
Subtidal Habitats (cont.)	
Algal Communities	
Substrate type	Almost all members of the macroalgal community are dependent on a hard substrate for holdfast attachment. If this substrate is lost, the community cannot exist at that location.
Replacement stock	Attempts have been made to restore economically important kelp beds using sporophytes grown in the laboratory and by transplanting adults. Associated environmental parameters and limitations should be analyzed prior to project initiation to determine if these project types are viable for a specific location.
Predation and competition	Attempts have been made to restore kelp beds by artificially reducing predation and competition. Manipulation to restore or rehabilitate an existing biological community should be investigated on a case-by-case basis.
Seagrass Beds	
Water quality	Water clarity controls the depth and species zonation of seagrass beds. Nutrient concentrations affect light reaching the seagrasses by promoting phytoplankton and epiphyte growth. Temperature controls seagrass latitudinal distribution as well as their growth and respiration rate. Salinity controls the species composition with respect to distances from freshwater inputs.
Wave energy	Wave action controls the distribution of seagrasses in near coastal waters. Near coastal seagrass beds occur only along very low energy coastlines.
Depth	The depth to which seagrass beds occur is a function of water clarity. Depth of existing seagrass communities in a given area is an environmental parameter that must be considered carefully before any seagrass restoration work is undertaken.
Donor species and planting strategy	If replanting is to be undertaken, care must be given to the selection of the donor species for repopulating a seagrass bed. Factors that must be considered in advance of any restoration work include the planting strategy in terms of spacing of planted cores, similarity of environmental conditions within the donor and recipient beds, and ability of the proposed donor bed to sustain removal of the projected amount of plant material.
Non-Vegetated Soft Bottom Communities	
Water quality	If ambient water quality conditions are altered, species composition in the non-vegetated soft bottom community is altered.
Substrate type	Infaunal community distribution within the non-vegetated soft bottom habitat is largely determined by sediment size. If the sediment size in a given location is altered, the infaunal community at that location is altered.

TABLE 5A-3. (cont.)

Parameter	Comment
Subtidal Habitats (cont.)	
Depth	If the depth of the seafloor is permanently changed either through the piling up of spoil material or the creation of borrow pits, the infaunal community inhabiting these habitats may change.
Substrate chemistry	Toxic or other chemical substances within offshore dredge spoil deposits may prevent the reestablishment of the soft bottom infaunal community.

TABLE 5A-4. RESTORATION PROJECTS ALONG OPEN COASTLINE AND NEAR COASTAL WATERS

Problem	Possible Solutions
Beach and Intertidal Habitats	
Rocky Shorelines	
Habitat deterioration or loss caused by modifications in water flow patterns, tidal range, water quality, or mechanical destruction.	Hydrological and water quality manipulations: Return the system to natural range of water quality conditions, flow patterns, or other conditions. Substrate manipulation: Replace or repair the original hard substrate.
Sandy Beaches and Sand Dunes	
Beach and dune habitat loss caused by natural erosion or erosion induced by channel and inlet maintenance.	Substrate manipulation: Replace substrate by pumping sand from offshore. Substrate manipulation: Mechanically reconstruct sand dunes.
Subtidal Habitats	
Coral Reefs and Live Bottom Areas	
Coral reef and live bottom habitats altered or destroyed as the result of water quality deterioration or mechanical damage (e.g., ship groundings, anchoring).	Water quality manipulation: Return the system to natural range of water quality conditions. Substrate manipulation: Mechanically reconstruct destroyed or damaged habitat.
Worm Rock Reefs	
Worm rock habitat altered or destroyed by hydrodynamic changes, water quality deterioration, or mechanical damage.	Hydrological and water quality manipulation: Return the system to natural range of water quality conditions, water flow, or other conditions. Substrate manipulation: Return the habitat to original or natural conditions in terms of sediment particle size and composition. Biological manipulation: Transplant fragments of sabellariid colonies into appropriate areas to speed recolonization. Substrate manipulation: Place hard substrate in the surf zone to act as substrate for worm colony attachment.
Artificial Reefs	
Artificial reefs mechanically damaged, buried, or sunk into soft bottom substrate.	Substrate manipulation: Position new artificial reef substrate in essentially the same location.

TABLE 5A-4. (cont.)

Problem	Possible Solutions
Algal Communities	
Algal communities altered or destroyed as the result of water quality deterioration or mechanical damage.	Water quality manipulation: Return the system to natural range of water quality conditions. Substrate manipulation: Mechanically reconstruct destroyed or damaged habitat. Biological manipulation (kelp beds): Remove kelp grazing sea urchins (species control), thin the understory of possible plant competitors, transplant adults, and seed areas with sporophytes grown in the laboratory.
Seagrass Beds	
Seagrass beds destroyed because of water quality deterioration or mechanical damage (e.g., dredge-and-fill projects).	Water quality manipulation: Return the system to natural range of water quality conditions. Substrate and hydrological manipulation: Mechanically stabilize the eroding edge(s) of seagrass bed, allowing the seagrass bed to reform in the damaged area. Also requires controlling water flow. Biological stocking: Replant damaged or destroyed seagrass beds. Typically this is done using cores or living sections of seagrass collected from an adjacent seagrass bed. This technique is similar to "sprigging" in terrestrial habitats.
Non-Vegetated Soft Bottom Communities	
Communities destroyed or damaged by dredging, burial with dredge spoil, and/or water quality deterioration.	Water quality manipulation: Return the system to natural range of water quality conditions. Substrate manipulation: Cap spoil deposits with natural sediments from the environment. Substrate manipulation: Fill borrow pits to recreate original bottom contours. Substrate manipulation: Remove or cap spoil deposits containing toxic substances.

reintroduction of endangered and threatened species or species of special concern to the newly created beach dune habitat may prove a viable method of rehabilitating this community.

Habitats that are particularly sensitive to deteriorating water quality are coral reefs and live bottom areas, seagrass beds, and algal communities. Where water quality problems such as turbidity, nutrients, or toxic contaminants have caused or contributed to habitat loss, management actions are needed to improve water quality or restoration/rehabilitation projects will ultimately fail.

Mechanical damage to coastal and nearshore habitats often results from either accidents (e.g., ship groundings on a coral reef) or from deliberate actions in a specific location (e.g., dredging). Water quality manipulations or other management actions may not be as important in such restoration projects because the event that caused the damage is controllable or not likely to be repeated.

Three case studies are presented below to illustrate restoration projects in open coastline and near coastal waters. The first case study describes efforts to restore the structural stability of coral reefs damaged by shipwrecks within the Florida Keys National Marine Sanctuary. Creation of an artificial reef is described in the second case study. This effort was conducted as a mitigation measure to compensate for anticipated habitat losses resulting from a beach nourishing project along the shoreline in Boca Raton, Florida. The last case study illustrates a project designed to restore the structure and benthic communities of a beach habitat changed during the period of several decades by the development of park facilities.

Coral Reef Restoration of Shipwreck Sites within Florida Keys National Marine Sanctuary

On October 25, 1989, the *M/V Alec Owen Maitland*, a 155-ft oil supply vessel, ran aground in the Florida Keys National Marine Sanctuary just south of Carysfort Reef in the Upper Florida Keys. Only 3 weeks later, the *M/V Elpis*, a 470-ft freighter, ran aground in the sanctuary just 4 miles to the south at Elbow Reef. The locations of these sites within the Florida Keys National Marine Sanctuary is shown in Figure 5A-8.

Reef damage at the *M/V Maitland* site consisted of two large craters, or "blowholes," in the reef crust that were caused by the ship's screws while trying to free the vessel. Damage at the *M/V Elpis* site was more widespread and in deeper water. As the *M/V Elpis* entered the reef tract, her hull clipped the tops from a number of coral heads and crushed the underlying reef. Again, two large craters were created by the ship's screws as she tried to free herself. In both cases, once

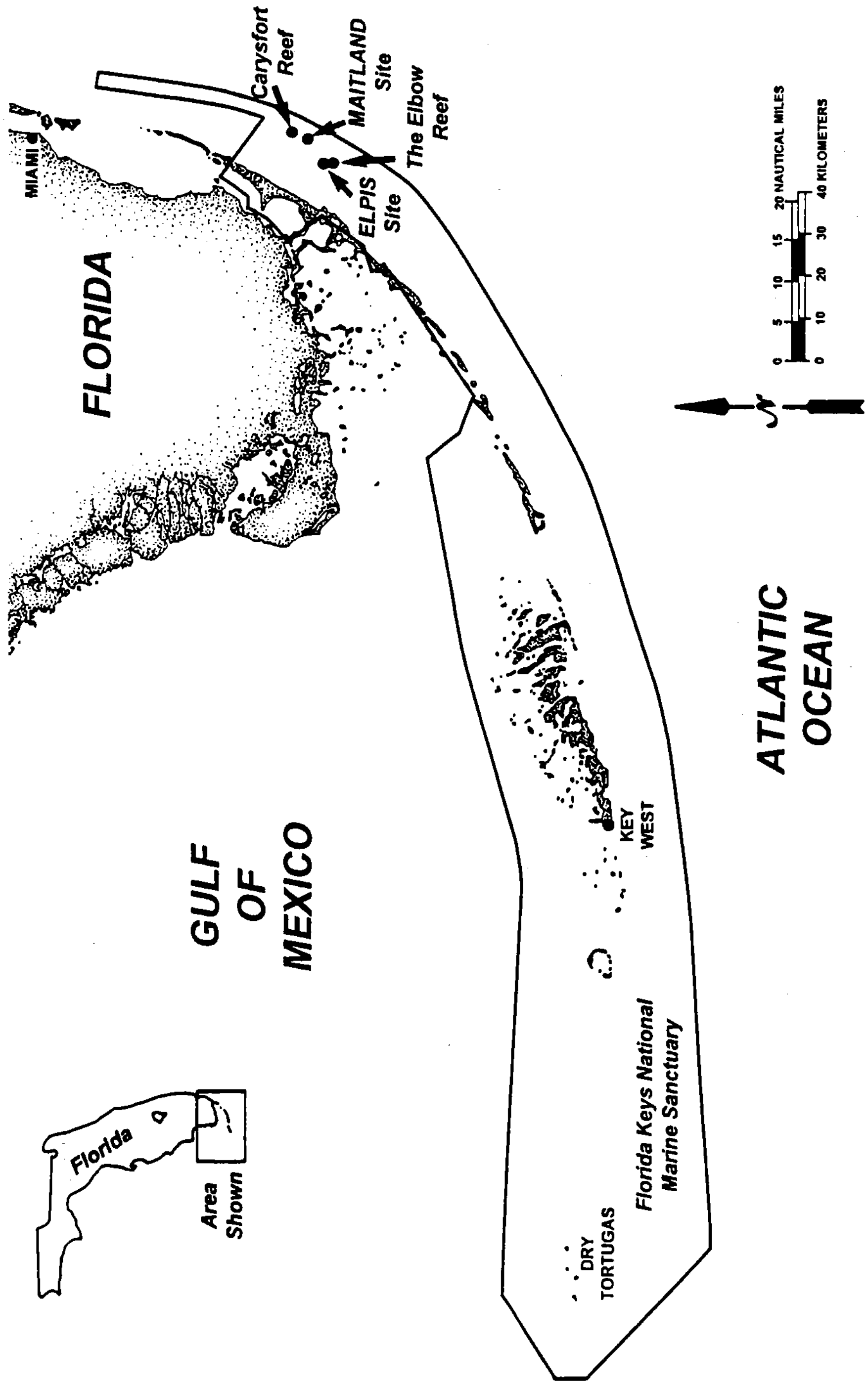


FIGURE 5A-8. Location of *M/V Maitland* and *M/V Elpis* coral reef restoration projects.

the structural integrity of the coral reef was breached, erosion began to proceed rapidly around the craters left by the grounded vessels.

In 1993, the National Oceanic and Atmospheric Administration initiated efforts for the structural restoration of the two grounding sites under the authority of the National Marine Sanctuaries Act. Funding for this restoration effort came from monetary assessments levied against the vessel owners. By federal law, money collected in such settlements must be used to restore the injured sites or to mitigate for the incurred injuries through environmental restoration of similar sites. The goals of this restoration project were to:

- Restore structural stability to the reefs and prevent further erosion
- Match as closely as possible the pre-grounding conditions of the reefs and provide suitable habitat for the natural colonization of corals and fishes
- Cause no additional damage to the surrounding sanctuary reefs.

Restoration Approach

Reconstruction of the reefs began with a detailed bathymetric survey of the damaged sites. Diver measurements were combined with other survey data to create 3-dimensional plots of the grounding areas to evaluate various design alternatives. Design alternatives were evaluated using a decision matrix based on project goals and final approved restoration plans.

Restoring the craters caused by the *M/V Elpis* involved removing large reef fragments from the pits, pushing the surrounding rubble berm into the pits, and adding limestone boulders from offsite. Carbonate sand was used to fill the crevices in the reef material, and the filled pit was sealed by replacing the reef boulders that had been previously removed. This plan removed the most unstable rubble from the reef surface area and exposed the larger, more irregular segments of reef rock in a stable matrix for colonization by the biological community.

Restoring the site of the *M/V Maitland* disturbance was more difficult because of the very shallow depth of the reef. Before the restoration project was started, the original two blowholes had been enlarged by wave erosion to form one major crater. The fragile reef perimeter of this crater was continually collapsing due to fracturing by waves and currents. Restoration design called for the placement of 40 pre-fabricated "Reef Replacing Armor Units" in the crater. These units were then surrounded with a specially formulated underwater cement to bind them to the coral perimeter and to fill the gaps between the units. Exposed cement areas were covered with reef limestone material to facilitate settling by normal reef biota.

Evaluation of Restoration Efforts

The primary goals of this restoration project were to correct area-specific damage and prevent further erosion of a Florida coral reef ecosystem. Restoration of a viable coral reef community at the specific sites where reef damage had occurred was to be accomplished by stabilizing the habitat and allowing natural recolonization to take place.

The scoping process for this project included an extensive analysis of the coral and coral reef material in the surrounding areas of coral reef growth. Precise measurements were made of the craters to be filled. The "Reef Replacing Armor Units" were designed and modified based on these studies. Constant monitoring and feedback during both the design and installation process among all groups involved (regulatory agencies, design engineers, marine biologist, and construction crews) allowed critical decisions to be made in the field in "real time" and with all pertinent data available to all critical parties.

All the primary goals of this project were accomplished. The restored reef surface area should naturally reach an ecological equilibrium with the surrounding reef habitat. Continued monitoring of the restored sites is being conducted by Florida Keys National Marine Sanctuary personnel.

For more information on this project, contact:

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Boca Raton Artificial Reef Project

In the late 1980s, the City of Boca Raton, Florida, in conjunction with the U. S. Army Corps of Engineers (Corps), conducted a major beach nourishing program along large areas of its shoreline. This program was conducted in several incremental phases. All phases of the program were designed so as not to cover any nearshore beach rock habitat during restoration of the target beach; however, it became apparent after the Phase 1 project that littoral sand transported from the nourished area did impinge upon, and in some cases cover completely, well-established, down-current rock outcrops.

Based on the footprint of the proposed Phase 2 element of this beach nourishment program, in 1987, it was projected that there would be significant sand inundation of Red Reef Rock, located south of the Spanish River Park area in Boca Raton

(Figure 5A-9). Red Reef Rock was, at that time, a favorite snorkeling spot for parkgoers, as well as a significant long-term beach rock habitat that supported a nearshore biological community. After considerable public controversy, the Boca Raton Artificial Reef Project was proposed as up-front mitigation for anticipated habitat losses. The artificial reef was installed in April and May of 1988, and the proposed beach nourishment project took place in June and July of that year.

The goals of this mitigation effort, sponsored jointly by the City of Boca Raton and the Corps, were to:

- Develop a viable, hard-bottom habitat to mitigate anticipated sand inundation of hard-bottom habitat on the northern side of Red Reef Rock
- Install this artificial hard-bottom habitat prior to the actual beach nourishment project
- Provide a new snorkeling area for city residents to take the place of the Red Rock Reef area to be lost.

Restoration Approach Used

The artificial reef consisted of six limestone boulder modules placed between 100 and 150 ft from shore, approximately 1 mile south of Spanish River public park in Boca Raton. As designed, these artificial reef modules consisted of 20–30 limestone boulders cemented together in two layers. The lower layer included an average of 16 boulders per module, and the upper layer included an average of 9 boulders per module. As constructed, the modules weighed between 3 and 5 tons, had a footprint of approximately 20 ft^2, and had a vertical relief of 3–4 ft. The modules were placed in an area where a thin sand veneer covered an underlying hard bottom layer of Anastasia limestone. This positioning was designed to prevent the modules from settling into the sand and losing their usefulness as hard-bottom habitat.

Monitoring began as soon as the modules were in place (May 1988) and continued regularly for 1 year after the completion of beach nourishment project. Monitoring was continued on a less regular basis after this period.

Evaluation of Restoration Efforts

Final estimates of the hard-bottom area inundated by littoral sand transported to the Red Rock Reef area ranged from 5 to 10 acres. Based on acreage alone, the compensation ratio on the restoration project was extremely low. One argument, made at the time the reef modules were originally installed, was that the habitat created had considerably greater vertical relief than the inundated beach rock and,

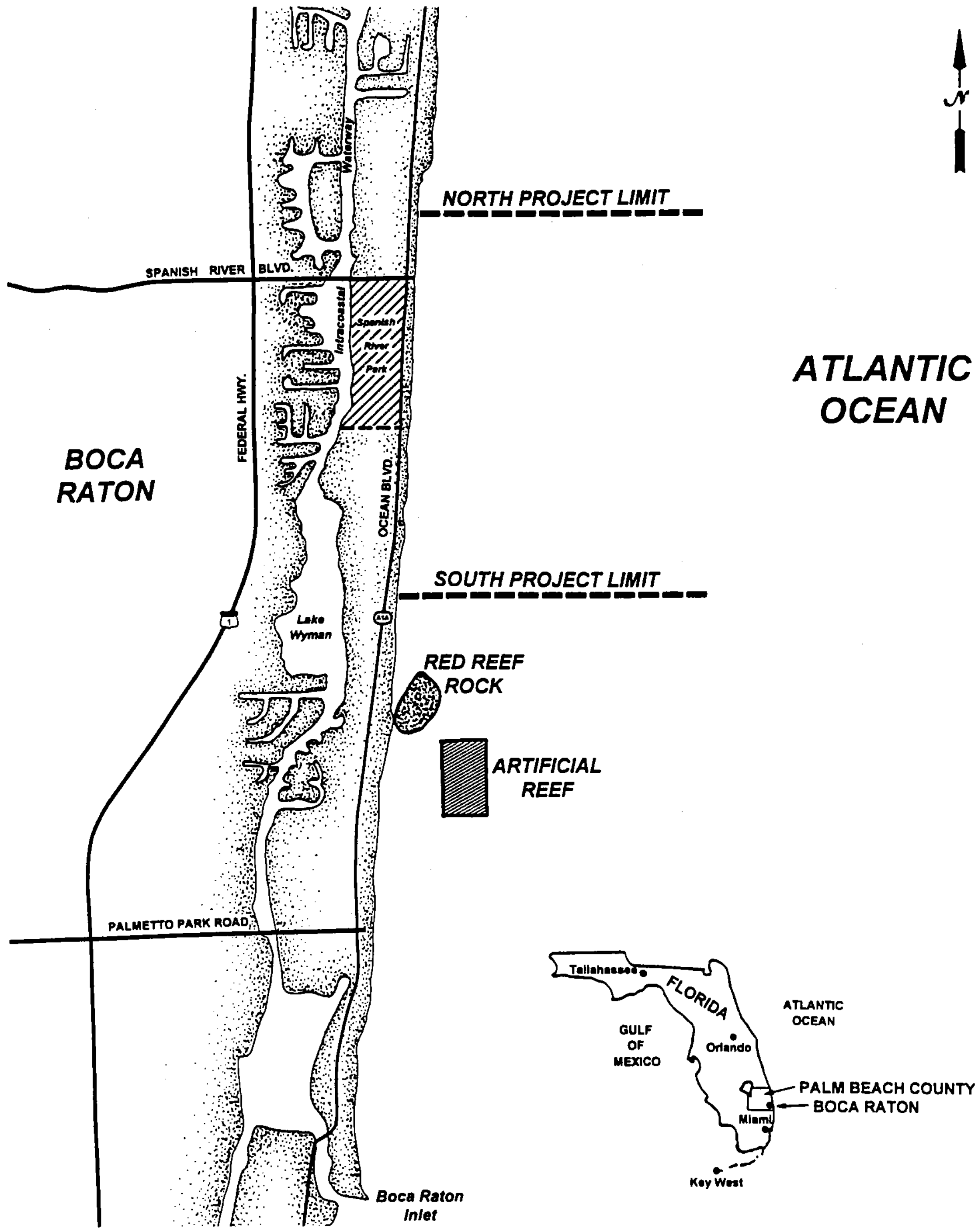

FIGURE 5A-9. Location of the Boca Raton artificial reef area.

thus, was potentially of more ecological value. Subsequent monitoring was unable to substantiate this claim. Furthermore, the created habitat did not support the same community as the degraded habitat. The placement of the artificial reef modules in approximately 6 ft of water also greatly reduced their attractiveness to novice snorkelers who had been visiting Red Rock Reef, which was found in shallower water. Finally, recent site visits indicate that two of the six modules have settled into the sand and have lost much of their usefulness as artificial habitat; the remaining four are still viable.

An adaptive management strategy would have improved this restoration project. Because this project was a first attempt to mitigate for lost nearshore beach rock habitat, several types of artificial reef modules could have been placed and the developing communities monitored to see which type most nearly approximated the one lost. Review of the snorkeling use patterns of parkgoers would have shown project planners the need to place at least some of these artificial habitats in shallower (3 ft or less) water. While this project did take an ecosystem or habitat approach toward restoration, design and installation of several different types of artificial habitats would have helped ensure success for the goals of reproducing both the community and the human use pattern of the area degraded.

This project was important because it established the principle that nearshore beach rock hard-bottom communities were a significant part of the coastal ecosystem and that their loss required mitigation. The Boca Raton Artificial Reef Project also established the precedent for up-front or prior mitigation before permits for beach nourishment projects would be issued.

Whether or not this project accomplished its goals is a matter of debate, but there is no question that it was a ground-breaking project in terms of establishing mitigative requirements and permitting procedures. Today, this type of prior mitigation for anticipated losses in the nearshore beach rock community is common along the southeast Florida coast. In addition, required ratios of habitat created vs. habitat lost are now much higher than they were at the time of the Boca Raton Artificial Reef Project.

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Lincoln Park Beach Shoreline Erosion Control Project

Lincoln Park was created in 1922 when the City of Seattle, Washington, acquired 130 acres of land along Puget Sound at Williams Point (Figure 5A-10). The park was opened to the public in 1925, and the majority of development took place in the 1930s. In 1936, a cobblestone and mortar seawall was constructed along the Lincoln Park shoreline to protect the newly constructed park facilities. As a result of this beach armoring (seawall construction), wave action gradually changed the beach habitat structure at Lincoln Park from a mixed cobble/coarse sediment habitat to a gravel habitat, and, eventually, to a hardpan habitat. The biological community likewise shifted from a relatively sparse community of intertidal detritivores to a community dominated by sessile macrofauna and seaweeds. Twice during the 1950s, large sections of the seawall failed as the result of undermining of the seawall toe by wave action. Repairs to the seawall, as well as backfilling and repairing of the adjoining asphalt walkways and service roads, were frequently required throughout the 1960s and 1970s. Despite these stopgap measures, erosion problems persisted as the seawall aged. In 1981, winter storm waves dislodged a 90-ft section of the seawall at Williams Point, in addition to breaking through the wall and washing out backfill at several points along the adjoining beach area.

In 1983, the Corps Seattle District evaluated two long-term plans for controlling beach erosion at Lincoln Park. One of these plans involved a concrete and sheet-pile barrier for seawall toe protection. The other plan called for restoring the original cobblestone and coarse sediment beach by placing fill material directly in front of the seawall. The latter alternative was selected, and initial beach restoration was completed in 1988.

Key components of this erosion protection plan included periodic renourishment and revetment stabilization to maintain the restored beach and extensive physical and biological monitoring to quantify what changes in intertidal habitats and biota could be expected from changes in the beach structure. The first renourishment effort took place in 1994.

The goals of the Lincoln Park Shoreline Erosion Control Project, sponsored by the City of Seattle and the Corps Seattle District, were to:

- Protect the seawall, access roads, and pedestrian walkways at Lincoln Park
- Provide quantitative data on the physical impacts of both historic seawall armoring and beach nourishment
- Provide long-term environmental data on the effects of beach nourishment and subsequent redistribution of fill materials in nearshore habitats.

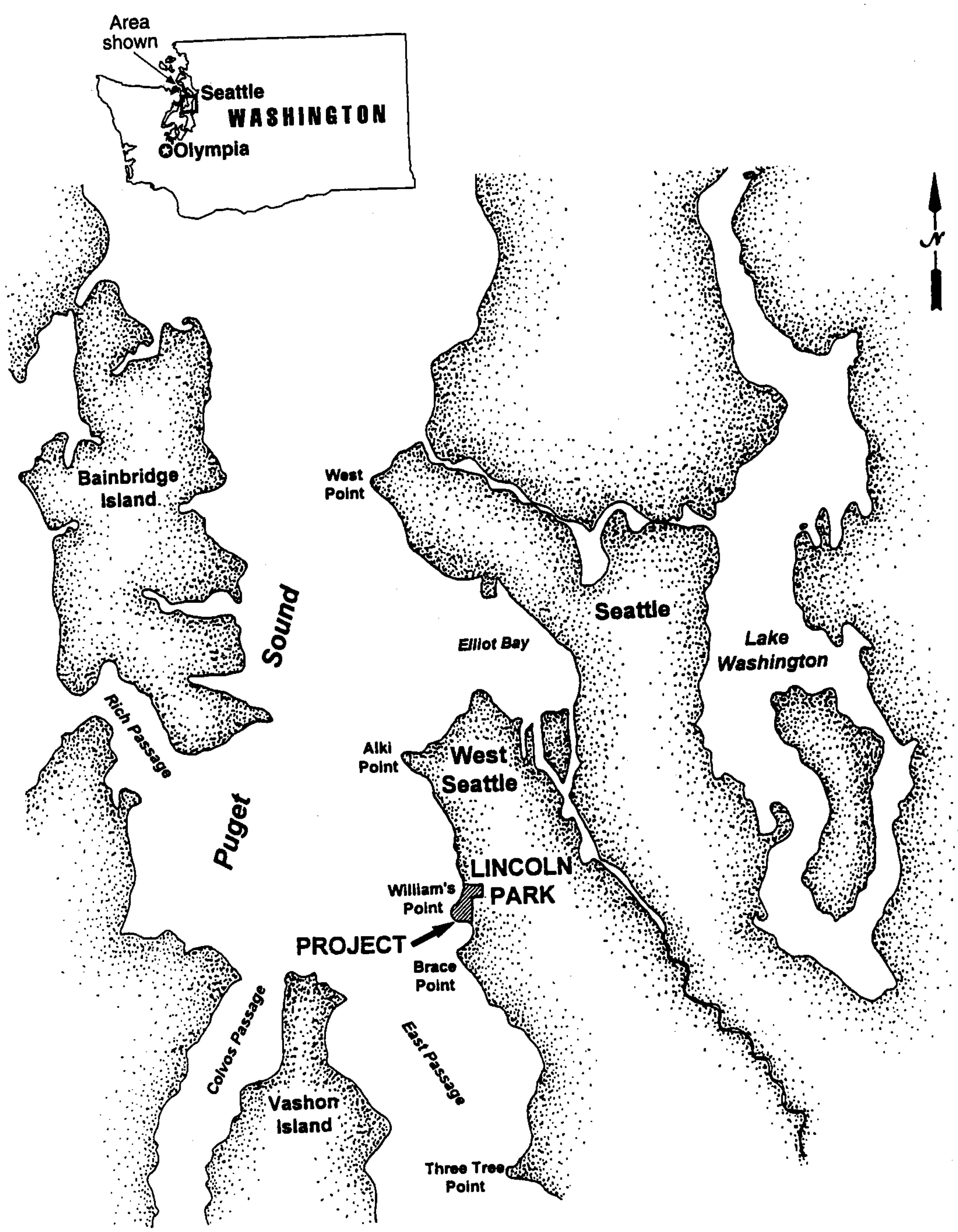

FIGURE 5A-10. Location of the Lincoln Park erosion control project.

Restoration Approach

In 1988, approximately 20,000 yd^3 of sand, gravel, and cobbles were placed on the beach. Additionally, 3,900 tons of armor stone were placed at Williams Point to replace sections of a failed seawall. After placing the sand gravel, and cobbles on the beach, these substrates were distributed and graded with earth-moving equipment at low tide. Placing these substrates in front of the existing seawall effectively negated the deleterious effects of the seawall on the littoral system. Benthic recovery studies and biological monitoring data, as well as beach profiles, were used to help design the 1994 renourishment project.

Evaluation of Restoration Efforts

As adopted, the plan for the Lincoln Park Shoreline Erosion Control Project illustrates a long-term adaptive management strategy. It also illustrates ecosystem-level planning, in that it attempted to restore an intertidal ecosystem that had vanished as a result of beach armoring. The cobble beach restoration efforts were aimed at restoring community-level, rather than species-specific, habitats. The project was designed to allow the newly restored cobble beach to reach an ecological equilibrium with the surrounding nearshore habitats of Puget Sound. Monitoring data collected after the 1988 beach restoration were used in preparing the subsequent renourishment plan for the 1994 project, establishing a feedback loop to help evaluate, refine, and achieve project goals.

Based on the results to date, the Lincoln Park Shoreline Erosion Control Project appears to be accomplishing its stated goals. Since placement of the original beach nourishment material in 1988, drift logs and other debris have accumulated on the first 8–10 ft of the beach directly seaward of the seawall. In addition to helping dissipate wave energy during high tide conditions and protecting the seawall toe, drift logs also serve to trap organic debris and fine sediments between them and the seawall. Prior to beach nourishment, wave energy in front of the seawall was too powerful to allow the accumulation of drift logs and organic debris.

As the Lincoln Park shoreline adjusts to a new equilibrium, beach sediments are mobilized and there is a certain amount of sediment transport to both north and south of the site. Monitoring through 1992 indicated that the Lincoln Park beach restoration was having no adverse affect on adjacent shorelines because the beach nourishment had not materially altered the transport processes in the area. The effect of the sediments that are transported offsite on the existing biological communities is still being monitored.

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REFERENCES

Andrews, H.L. 1945. The kelp beds of the Monterey region. Ecology 26:24-37.

Bohnsack, J.A. 1989. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? Bull. Mar. Sci. 44:631-645.

Bohnsack, J.A. 1991. Habitat structure and the design of artificial reefs. pp. 412-426. In: The Physical Arrangement of Objects in Space. S.S. Bell, E.D. McRoy, and H.R. Mushinsky (eds). Chapman and Hall, New York, NY.

Bohnsack, J.A., and D.L. Sutherland. 1985. Artificial reef research: a review with recommendations for future priorities. Bull. Mar. Sci. 38:11-39.

Coull, B.C. 1977. Ecology of marine benthos. University of South Carolina Press, Columbia, SC. 467 pp.

CSA. 1989. Southwest Florida nearshore benthic habitat study, narrative report. Prepared for the U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico Region, New Orleans, LA. Continental Shelf Associates, Inc., Jupiter, FL. 55 pp.

CSA and MLI. 1986. Florida Big Bend seagrass habitat study narrative report. Prepared for the U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, Metairie, LA. Continental Shelf Associates, Inc., Jupiter, FL, and Martel Laboratories, Inc. 47 pp. + app.

Cummings, S.L. 1990. Colonization of a nearshore artificial reef at Boca Raton (Palm Beach County), Florida. Thesis. Florida Atlantic University, Boca Raton, FL. 99 pp.

Dawson, E.Y., and M.S. Foster. 1982. Seashore plants of California. University of California Press, Berkeley, CA. 226 pp.

Doherty, P.J., and D. McB. Williams. 1988. The replenishment of coral reef fish populations. Oceanogr. Mar. Biol. 26:225-234.

Foster, M.S., and D.R. Schiel. 1985. The ecology of giant kelp forests in California: a community profile. U.S. Fish Wildl. Serv. Biol. Rep. 85(7.2). U.S. Fish and Wildlife Service, Washington, DC. 152 pp.

Foster, M.S., A.P. De Vogelaere, J.S. Oliver, J.S. Pearse, and C. Harrold. 1991. Open coast intertidal and shallow subtidal ecosystems of the Northeast Pacific. pp. 235–272. In: Intertidal and Littoral Ecosystems. Ecosystems of the World 24. A.C. Mathiesen and P.H. Nienhuis (eds). Elsevier, New York, NY. 564 pp.

Gallaway, B.J., and G.S. Lewbel. 1982. The ecology of petroleum platforms in the northwestern Gulf of Mexico: a community profile. FWS/OBS-82/27. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 92 pp.

Gilmore, R.G., J.C. Donahue, D.W. Cooke, and D.J. Herrema. 1981. Fishes of the Indian River Lagoon and adjacent waters, Florida. Harbor Branch Foundation Technical Report No. 41. 36 pp.

Gore, R.H., L.E. Scotto, and L.J. Becker. 1978. Community composition, stability, and trophic partitioning in decapod crustaceans inhabiting some subtropical sabellariid worm reefs. Bull. Mar. Sci. 28(2):221–248.

Goreau, T.F., and N. Goreau. 1960. The uptake and distribution of carbon in reef building corals with and without zooxanthellae. Science 131:668–669.

Grant, J.J., K.C. Wilson, A. Grover, and H.A. Togstad. 1982. Early development of Pendleton artificial reef. Mar. Fish. Rev. 44(6-7):53–60.

Gray, J.S. 1981. The ecology of marine sediments. Cambridge University Press, New York, NY. 185 pp.

Gross, M.G. 1993. Oceanography. Prentice-Hall, Englewood Cliffs, NJ. 446 pp.

Jaap, W.C. 1983. The Florida Keys and eastern Gulf of Mexico reef and "live bottom" communities including the Florida Middle Ground. In: Conference Proceedings: Gulf of Mexico Trends for the 80's. Tulane University, June 23–27, 1983.

Jaap, W.C. 1984. The ecology of the south Florida coral reefs: a community profile. FWS/OBS-82/08. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 138 pp.

Jaap, W.C., and P. Hallock. 1990. Coral reefs. pp. 97–116. In: Synthesis of Available Biological, Geological, Chemical, Socioeconomic, and Cultural Resource Information for the South Florida Area. OCS Study MMS 90-0019. U.S. Department of the Interior, Minerals Management Service, Atlantic OCS Region, Herndon, VA.

Jackson, J.B.C. 1979. Morphological strategies of sessile animals. Systematics Association Special Volume No. 11:499-555. Academic Press, New York, NY.

Kaufman, W., and O. Pilkey. 1979. The beaches are moving. Anchor Press, Garden City, NY. 326 pp.

Kirtley, D.W. 1966. Intertidal reefs of Sabellariidae (Annelida:Polychaeta) along the coast of Florida. Thesis. Florida State University, Tallahassee, FL. 104 pp.

Kirtley, D.W. 1974. Geological significance of the polychaete annelid family Sabellaridae. Dissertation. Florida State University, Tallahassee, FL. 270 pp.

Komar, P.D. 1976. Beach processes and sedimentation. Prentice-Hall, Englewood Cliffs, NJ. 429 pp.

Lewis, J.R. 1964. The ecology of rocky shores. English University Press, London. 323 pp.

Littler, M.M., D.S. Littler, S.N. Murray, and R.S. Seapy. 1991. Southern California rocky intertidal ecosystems. pp. 273-296. In: Intertidal and Littoral Ecosystems. Ecosystems of the World 24. A.C. Mathiesen and P.H. Nienhuis (eds). Elsevier, New York, NY. 564 pp.

Marszalek, D.S., G. Babashoff, Jr., M.R. Noel, and D.R. Worley. 1977. Reef distribution in South Florida. pp. 223-229. In: Proceedings, Third International Coral Reef Symposium, Miami, FL.

Mathiesen, A.C., C.A. Penniman, and L.G. Harris. 1991. Northwest Atlantic rocky shore ecology. pp. 109-191. In: Intertidal and Littoral Ecosystems. Ecosystems of the World 24. A.C. Mathiesen, and P. H. Nienhuis (eds). Elsevier, New York, NY. 564 pp.

MRRI. 1984. South Atlantic OCS area living marine resources study, Phase III. Report to the U.S. Department of the Interior, Minerals Management Service, Washington, DC. Contract No. 14-12-0001-29185. South Carolina Wildlife and Marine Resources Department, Marine Resources Research Institute.

Multer, H.G. 1969. Carbonate rock environments. Miami Geological Society, Miami, FL. 152 pp.

NOAA. 1985. National artificial reef plan. NOAA Technical Memorandum NMFS OF-6. National Oceanic and Atmospheric Administration, Washington, DC. 70 pp.

Nybakken, J.W. 1988. Marine ecology: an ecological approach. Second edition. Harper & Row, New York, NY. 514 pp.

Odum, H.T., and B.J. Copeland. 1974. A functional classification of the coastal ecological systems. pp. 5–85. In: Coastal Ecological Systems of the United States. H.T. Odum, B.J. Copeland, and E.A. McMahan (eds). The Conservation Foundation, Washington, DC.

Odum, H.T., B.J. Copeland, and E.A. McMahan. 1974. Coastal ecological systems of the United States. Volume I. The Conservation Foundation, Washington, DC. 533 pp.

Ogden, J.C., and J.P. Ebersole. 1981. Scale and community structure of coral reef fishes: a long term study of a large artificial reef. Mar. Ecol. Prog. Ser. 4:97–103.

Parker, R.H. 1960. Ecology and distributional patterns of marine macroinvertebrates, northern Gulf of Mexico. pp. 302–337. In: Recent Sediments, Northwest Gulf of Mexico. American Association Petroleum Geologists.

Parker, R.O., Jr., D.R. Colby, and T.D. Willis. 1983. Estimated amount of reef habitat on a portion of the U.S. South Atlantic and Gulf of Mexico continental shelf. Bull. Mar. Sci. 33:935–940.

Petersen, C.G.J. 1918. The sea bottom and its production of fish food. A survey of the work done in connection with the valuation of Danish waters from 1883–1917. Rap. Danish Biol. Sta., Vol. 25. 62 pp.

Phillips, R.C. 1984. The ecology of eelgrass meadows in the pacific northwest: a community profile. FWS/OBS/84/24. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 85 pp.

Porter, J.W., L. Muscatine, Z. Dubinski, and P. Falkowski. 1984. Primary production and photoadaptation in light and shade-adapted colonies of the symbiotic coral, *Stylophora pistillata*. Proc. R. Soc. Lond. 222B:161–180.

Pritchard, P.C. 1979. Encyclopedia of turtles. Neptune: TFH Publ. Inc., Neptune, NJ.

Pulley, T. E. 1952. A zoogeographic study based on the bivalves of the Gulf of Mexico. Dissertation. Harvard University, Cambridge, MA.

Rabalais, N.N., and D.F. Boesch. 1987. Dominant features and processes of continental shelf environments of the United States. pp. 71–147. In: Long-Term Environmental Effects of Offshore Oil and Gas Development. D.F. Boesch, and N.N. Rabalais (eds). Elsevier, New York, NY.

Rezak, R., T.J. Bright, D.W. McGrail. 1985. Reefs and banks of the northwestern Gulf of Mexico: their geological, biological, and physical dynamics. John Wiley & Sons, New York, NY. 259 pp.

Rosenthal, R. J., W. D. Clarke, and P. K. Dayton. 1974. Ecology and natural history of a stand of giant kelp, *Macrocystis pyrifera*, off Del Mar, California. Fish. Bull. 72(3):670–684.

Rudolph, H.D. 1977. A taxonomic study of the polychaetous annelids associated with worm rock reefs of *Phragmatopoma* (Kinberg 1867) in Palm Beach County, Florida. Thesis. Florida Atlantic University, Boca Raton, FL. 304 pp.

Sandifer, P.A., J.V. Miglarese, D.R. Calder, J.J. Manzi, and L.A. Barclay. 1980. Ecological characterization of the Sea Island coastal region of South Carolina and Georgia: Volume III, biological features of the characterization area. U.S. Fish and Wildlife Service, Washington DC. 620 pp.

Sheehy, D.J. 1982. The use of designed and prefabricated artificial reefs in the United States. Mar. Fish. Rev. 44(6–7):4–23.

Shinn, E.A., B.H. Lidz, J.L. Kindinger, J.H. Hudson, and R.B. Halley. 1989. A field guide: reefs of Florida and the Dry Tortugas. U.S. Geological Survey, St. Petersburg, FL.

Smith, F.G. 1972. Atlantic reef corals. University of Miami Press, Coral Gables, FL. 164 pp.

Smith, G.B. 1976. Ecology and distribution of eastern Gulf of Mexico reef fishes. Florida Marine Research Publ. No. 19. Florida Department of Natural Resources, Marine Research Laboratory, St. Petersburg, FL. 78 pp.

Steele, J.H. 1974. The structure of marine ecosystems. Harvard University Press, Cambridge, MA. 128 pp.

Stephenson, T.A., and A. Stephenson. 1972. Life between tidemarks on rocky shores. W.H. Freeman, San Francisco, CA. 425 pp.

Taylor, W.R. 1979. Marine algae of the eastern tropical and subtropical coasts of the Americas. University of Michigan Press, Ann Arbor, MI. 870 pp.

Thayer, G.W., W.J. Kenworthy, and M.S. Fonseca. 1984. The ecology of eelgrass meadows on the Atlantic coast: a community profile. FWS/OBS-84/02. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 147 pp.

Thorson, G. 1957. Bottom communities (sublittoral or shallow shelf). pp. 461–534. In: Treatise on Marine Ecology and Paleoecology, Vol. I. Geol. Soc. Am. Mem. 67.

Valentine, J.W. 1963. Biogeographic units as biostratigraphic units. Bull. Amer. Petroleum Geol. 47(3):457–466.

Van Montfrans, J. 1981. Decapod crustaceans associated with worm rock (*Phragmatopoma lapidosa* Kinberg) in southeastern Florida. Thesis. Florida Atlantic University, Boca Raton, FL. 290 pp.

Wilson, K.C., R.D. Lewis, and H.A. Togstad. 1990. Artificial reef plan for sport fish enhancement. Marine Resources Administrative Report No. 91-15. Marine Resources Division, Long Beach, CA. 76 pp.

Wood, D.A. 1989. Official list of endangered and potentially endangered fauna and flora in Florida. Florida Game and Fresh Water Fish Commission, Tallahassee, FL.

Zieman, J.C. 1982. The ecology of the seagrasses of south Florida: a community profile. FWS/OBS-82/25. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 158 pp.

Zieman, J.C., and R.T. Zieman. 1989. The ecology of seagrass meadows of the west coast of Florida: a community profile. U.S. Fish Wildl. Serv. Biol. Rept. 85(7.25). U.S. Fish and Wildlife Service, Washington, DC.

5B. SUBTIDAL ESTUARIES

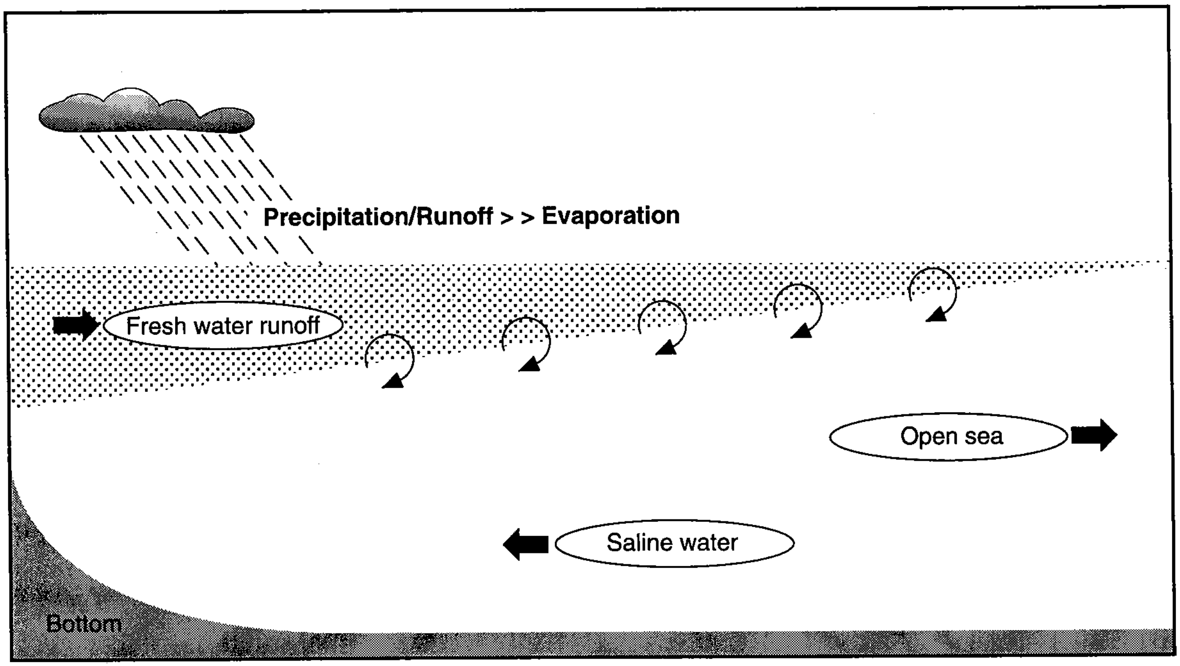
Donald Rhoads and John Lunz

Estuaries, where rivers and streams meet and mix with marine waters, comprise as much as 80–90 percent of the East and Gulf coasts of North America and as little as 10–20 percent of the steeper, Pacific coast of the United States (Emery 1967). Definitions of estuaries range from physical and geomorphological definitions in older references to hydrological and ecologically based definitions in more recent studies. Classical work on estuaries is largely based on northern European and North American estuaries, which are typically "normal" estuaries: estuaries where freshwater runoff is a significant dynamic force and drives saline water along the bottom toward the head of the estuary. The definition of a normal estuary given by Pritchard (1967) is: "An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with freshwater derived from land drainage." Freshwater runoff into estuaries leads to high sediment accumulation rates related to flocculation, which takes place within the mixing zone of relatively fresh and saline waters. Chesapeake Bay is frequently referred to as the "North American" type estuary and is held as an example of normal salinity circulation.

Less attention has been given to estuaries showing antiestuarine circulation, sometimes called "negative" estuaries or coastal lagoons. Such embayments exist in arid climates, where evaporation exceeds precipitation, causing hypersaline conditions to develop within the lagoon. Dense saline waters sink and move seaward along the bottom. Mass balance is maintained by normal salinity seawater moving into the lagoon along the surface (Groen 1969). Generic examples of normal and antiestuarine (lagoonal) circulation are shown on Figure 5B-1.

Because estuaries are located at the interface between land and sea, they are a zone of intense interaction between anthropogenic activities and natural processes. Estuaries afforded some protection from long fetch waves by barrier bars or headlands tend to accumulate fine-grained sediments that settle and accumulate in low kinetic energy environments. Virtually all sediment entering an estuary is ultimately trapped and stored within the bottom. Some of this sediment must be periodically removed by dredging to maintain navigation channels and berthing sites. A fraction of this fine-grained sediment may also be associated with contaminants potentially harmful to the estuarine ecosystem.

A. Normal Estuarine Circulation



B. Antiestuarine Circulation (Hypersaline Lagoon)

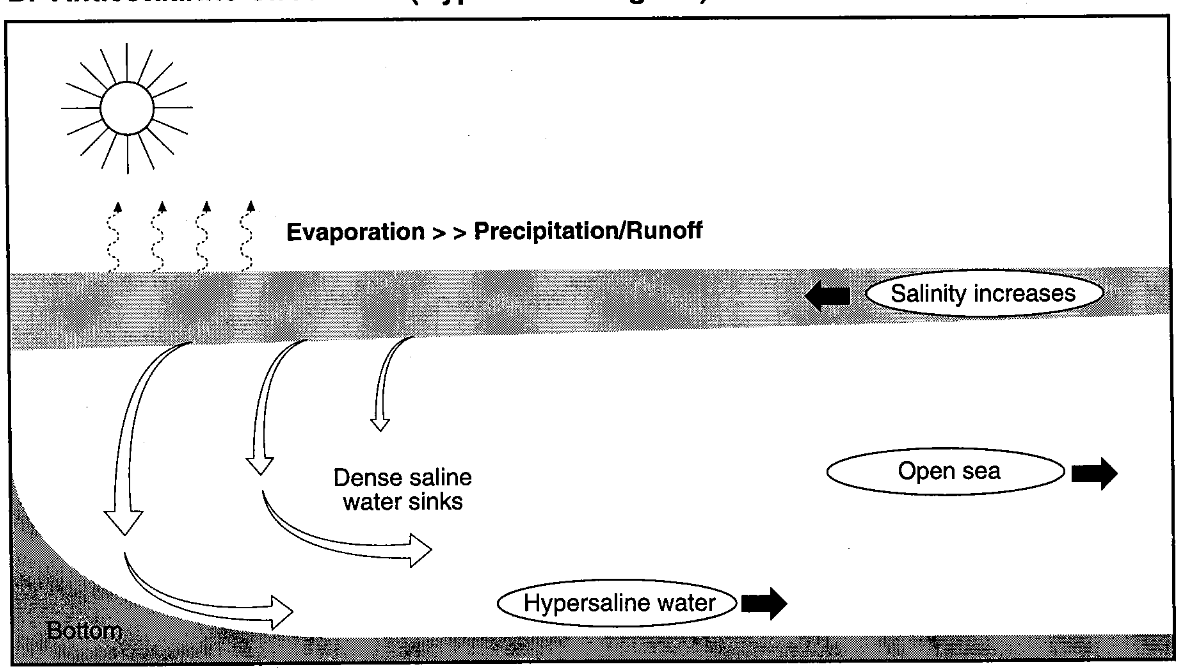


FIGURE 5B-1. Normal and antiestuarine (lagoon) circulation

ECOSYSTEM PROFILE

Coastal plain estuaries (e.g., Chesapeake Bay) are the best-studied type of estuary and are also called "drowned river valley estuaries." These estuaries formed as the post-Pleistocene sea level rose, inundating downstream portions of river valleys. Coastal plain salt marsh estuaries may lack a major river, but they have a well-defined tidal drainage network that dissects extensive coastal salt marshes with narrow tidal inlets connecting the system to the ocean; the estuary proper consists of tidal and drainage channels.

Lagoons (bar-built estuaries) are usually oriented with their major axis parallel to the coast, have a less well-drained subaqueous drainage channel network and are uniformly shallow, often less than 2-m deep over large areas. In arid climates, these lagoons may display antiestuarine circulation.

Fjords, which are glacially eroded river valleys drowned by a rising sea level, are deep-water estuaries. Typically, a shallow bar at the mouth of the estuary is the remnant of a terminal moraine that forms a sill varying in depth from 10–90 m; however, interior fjord depths can exceed 800 m. The length of the basin can extend for hundreds of kilometers inland. The typical cross-section is U-shaped as a result of glaciation. Therefore, steep fjord walls are nearly vertical, preventing sediment accumulation along the sides. Sediment does accumulate on the deep basin floor, which can become locally anoxic as the result of restricted circulation caused by the sill and/or water column stratification.

Tectonically caused estuaries are estuaries caused by faulting, graben formation, landslides, or volcanic eruptions. The best-known example of such an estuary is San Francisco Bay.

The estuarine benthic ecosystem is often subdivided into two types (i.e., hard and soft bottom habitats). Very fine sand, silt, and clay tend to dominate the estuarine bottom, and this facies, which is relatively level (i.e., flat), is referred to as soft bottom because the fine-grained sediments typically have high water content (>70 percent by weight) due to intensive bioturbation. Those parts of the estuarine floor exposed to wave and current scour lack fine-grained sediment cover and have hard bottom habitats (rock outcroppings, hard-packed sand and gravel, or biological structures such as reefs or oyster banks). Soft and hard bottom estuarine habitats have very different physical, chemical, and biological characteristics and can be separated into those parts of the bottom that receive sufficient light for plant growth and those parts too shaded to support plants. The soft bottom profile described here focusses on the non-vegetated parts of the habitat because estuarine submerged aquatic vegetation (SAV) is discussed in *Estuarine and Coastal Wetlands*. The hard bottom profile includes both vegetated and non-vegetated habitats.

Soft Bottom Habitats

Soft-bottom environments are characterized by relatively level (i.e., flat) areas of unconsolidated granular sediments consisting of uncompacted sand, silt, clay (mud), or mixtures of these end members. These bottom habitats tend to dominate all of the geomorphic types of estuaries described above.

Geographic Distribution

Coastal plain salt marsh estuaries, such as Chesapeake Bay, are common along the Atlantic coast, from Massachusetts to Florida. Fjords occur above 45° latitude along the formerly glaciated coasts of the Pacific Northwest, British Columbia, and Alaska (Fairbridge 1980). Tectonic estuaries are also best represented on the West coast because of local and regional tectonic activity. North American lagoons are best developed along the Gulf of Mexico, especially along the Texas coast (e.g., Laguna Madre).

Geographic differences in the range of sediment grain size are often pronounced among estuaries. In previously glaciated areas or areas of morainal deposits, such as the New England coast, a broad range of grain sizes are common (clay, silt, sand, gravel, cobbles, boulders). Estuaries in the Gulf coast are substantially finer grained, reflecting, in part, the extensive deposits of silt-sized loess throughout the Mississippi River watershed. Tectonically active coastal areas undergoing uplift tend to have coarser sediments than coasts along passive or subsiding continental margins.

Zonation within Habitats

Major environmental gradients within estuaries are defined by the tidal and/or salinity range. Sedimentation also subdivides the soft bottom estuary into different facies.

Tidal and Salinity Range Zonation—Major environmental gradients within estuaries may be defined by the tidal and salinity range. Day et al. (1989) define three major zones of an estuary: 1) tidal river zone, a fluvial zone characterized by lack of ocean salinity but subject to the tidal rise and fall of sea level; 2) mixing zone (estuary proper), characterized by a mixing water mass and existence of strong physical, chemical, and biotic gradients from the tidal river zone to the seaward location of a river mouth bar or ebb-tidal delta; and 3) nearshore turbid zone, between the mixing zone and the seaward edge of the tidal plume at full ebb tide.

The degree of vertical and horizontal salinity stratification depends on the relative strength of river flow, ratio of precipitation to evaporation, tidal flow, basin geometry, channel hydraulics (expressed as a Reynolds number), and wind mixing. Strongly stratified systems, where river flow dominates wind and tidal mixing, have steep horizontal and vertical gradients in isohaline lines and are called salt-wedge estuaries. The lower Mississippi River is an example of a salt-wedge estuary. Partially mixed estuaries result where tidal flow is sufficiently strong to minimize salinity stratification in parts of the system. Chesapeake Bay is an example of a partially mixed estuary. Vertically homogeneous estuaries are those where wind and tidal flows are sufficiently strong to eliminate all salinity stratification. The wider reaches of the Delaware and Raritan Bay estuaries represent vertically homogeneous estuaries. Horizontal salinity gradients may exist, but isohalines are vertical, or nearly so, at any given sampling location. Sectionally homogeneous estuaries have segments of the estuarine reach where there are no horizontal or vertical gradients in salinity. A given estuary may change classification as seasonal runoff and mixing intensity changes.

In those estuaries with well-developed horizontal gradients in salinity (salt-wedge, partially mixed, or vertically homogeneous), the range of salinity excursion within a segment of the estuary is a first-order ecological factor for determining the distribution of most plant and animal life and is a first-order factor to consider in restoration projects. The Venice Classification System (Table 5B-1) is commonly used to ecologically classify habitats according to the range of salinity excursion within different reaches of an estuary (i.e., salinity biofacies).

Sediment Zonation—Horizontal gradients in sediment type are related to the proximity to source areas such as rivers and tidal creeks, channels, and wave/current exposed reaches. The most coarse-grained sediment is usually located around the unvegetated perimeter of an estuary where surface waves interact with the bottom. Deep protected areas of an estuary are generally occupied by muddy silt-clay sediments. Depth-related sediment zonation is a common facies pattern in estuaries. If a major river enters an estuary, gradients in sediment type may be correlated with salinity gradients (Schubel 1971). Any anthropogenic modification of bottom depth, rate and volume of freshwater discharge, and/or exposure to waves is likely to have a significant effect on sediment facies zonation, so this relationship must be considered in restoration projects.

Sediments are also zoned vertically or stratigraphically and, as such, record past changes in the estuarine environment. In vertical sections, sediments may be

TABLE 5B-1. ECOLOGICALLY MEANINGFUL SALINITY RANGES BASED ON THE VENICE CLASSIFICATION SYSTEM (Hypersaline lagoons have been added)

Division of Estuary	Salinity Range (ppt)	Zones	Ecological Classification
River	0.5	Limnetic	Limnetic
Head	0.5–5.0	Oligohaline	Oligohaline
Upper Reaches	5–18	Mesohaline	
Middle Reaches	18–25	Polyhaline	True Estuarine
Lower Reaches	25–30	Polyhaline	
Mouth	30–40	Euhaline	Stenohaline; Euryhaline; Migrants
Arid Climate Lagoon	>40	Hypersaline	Hypersaline

Source: Modified from Carriker (1967).

physically zoned into layers deposited at different times. Physical layering may be related to differences in energy regime, sediment grain size, sorting, porosity, water content, lithology, or combinations of factors. Stratification, as observed in long sediment cores (much longer than 1 m) or in acoustic reflection profiles, can be used to determine the long-term dynamics of the bottom. If long-term habitat stability is important in a restoration project, a coring program may prove valuable for assessing this factor. For example, those parts of a silt-clay bottom underlain by, or intercalated with, layers of sand/shell/gravel are poor candidates for long-term stability because the long-term sediment record indicates that the bottom may be periodically affected by storm reworking and/or sedimentation events related to high runoff.

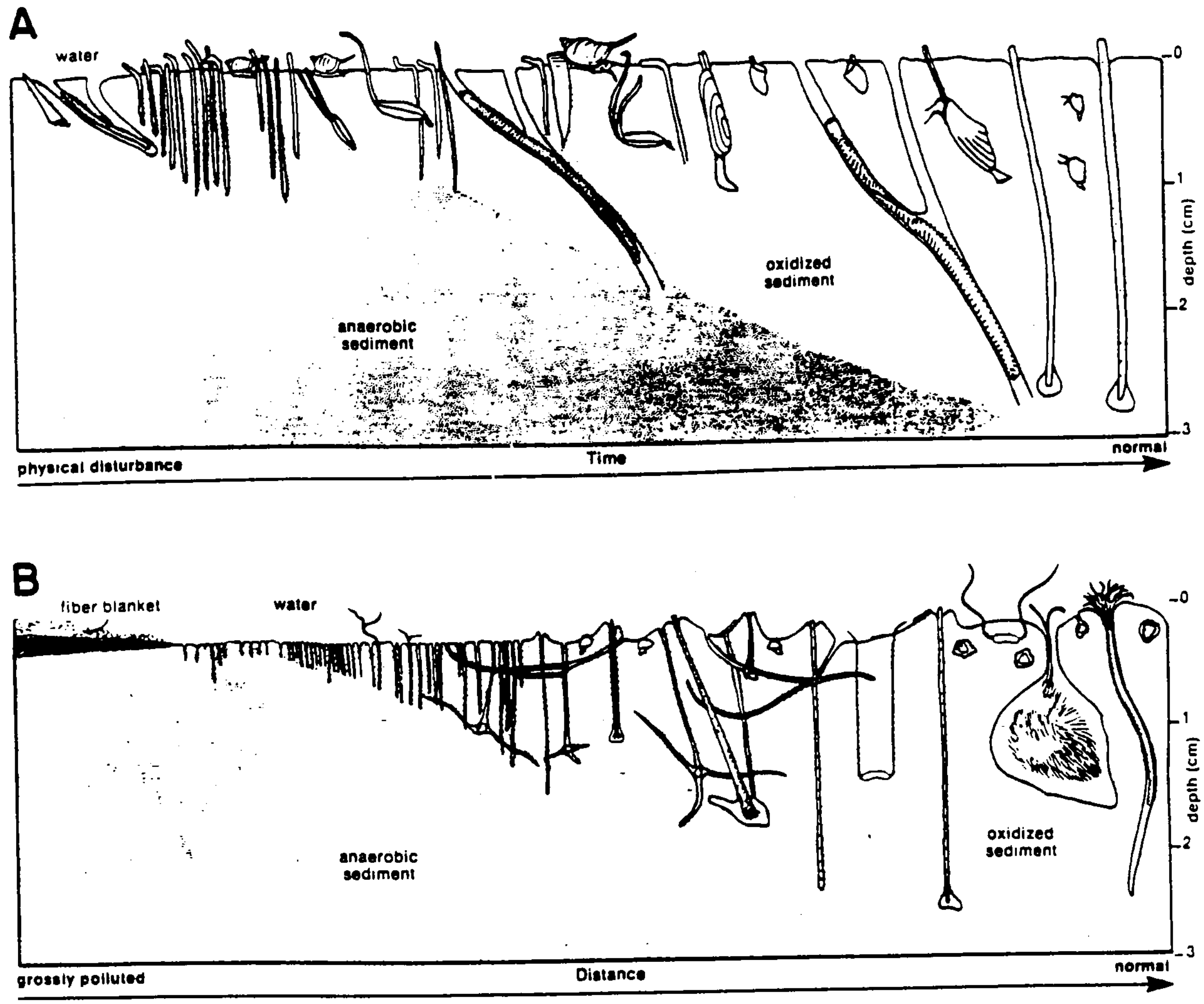
Sediments may also be vertically zoned as a result of chemical gradients extending through layers regardless of time of deposition. Vertical chemical gradients are related to the diffusion/advection of solutes and solids across the sediment-water interface. The sediment may represent a source term or consumption term for particular molecular species. Gradients exist in oxygen, sulfide, redox potential, pH, and/or dissolved materials (ions including salts, nutrients, organic matter, and metals). Of these chemical gradients, the redox gradient is one of the most important because it controls many biogeochemical cycles.

Chemical zonation in sediments is strongly tied to biological processes such as bioturbation, respiration, and decomposition by benthic infaunal invertebrates (Aller 1982). Most bioturbation takes place in the upper 10–15 cm of the bottom, but some deep burrowing shrimp (*Calianassa* spp.) can burrow to depths of 1 m or more (Aller 1982). From the perspective of habitat restoration, sediment stratification within the zone of biological penetration is important to consider. For example, capping of contaminated sediments by clean sediments (capping) is sometimes used to restore subtidal habitat quality. An important parameter in such a restoration project is the thickness of the cap that must be deposited to isolate buried contaminants from colonizing species over long periods of time. A long-term monitoring program must also be implemented to assure that the cap remains in place, especially following the passage of major storms. The vertical gradients in organic matter concentration and redox state may control benthic succession and, in turn, be modified by the same macrofaunal successional process (Rhoads and Boyer 1982; Forbes et al. 1994). Benthic algae and microorganisms (bacteria, fungi, protozoans) are important in benthic metabolism, and these components are also vertically zoned according to redox gradients (Meyers et al. 1987).

Biological Community

Estuarine biological communities can be described by their salinity tolerance (Carriker 1967), trophic relationships (Levinton et al. 1984), or successional dynamics (Pearson and Rosenberg 1978; Rhoads et al. 1978). Soft bottom communities are numerically dominated by organisms that live within the bottom (infauna). If the bottom is located above the photosynthetic compensation depth for plant growth, plants may serve as foundation species, affecting the distribution of a number of both epifaunal and infaunal species (see *Estuarine and Coastal Wetlands* section). In unvegetated habitats, a wide variety of infauna predominate, including microorganisms such as bacteria, fungi, and yeast cells; protozoa such as ciliates and foraminifera; permanent and temporary meiofauna living in interstitial pore spaces such as ostracods, tardigrades, and copepods; and early developmental stages of larger invertebrates (temporary meiofauna). Microorganisms and meiofauna serve important functions as processors of organic detritus (Christian and Wetzel 1978; Gerlach 1971). Macrofauna (>300 μm), such as polychaete or oligochaete worms, small crustaceans, echinoderms, and molluscs, form an important group of primary consumers, secondary consumers, and detritivores. The distribution of species within these major groups can be related to benthic disturbance (Figure 5B-2). The ecological attributes of disturbed benthic ecosystems appear to have some universal features, which makes it possible to qualitatively predict the consequences of disturbance and restoration (Table 5B-2). Predatory megafauna, including crabs and lobsters, burrowing fish (sand lance or garden eels), and demersal fish such as flounder and cod, form an important food chain link between macrofauna and humans. The nature and efficiency of this linkage is sensitive to the benthic successional paradigm shown in Figure 5B-2 and Table 5B-2. Epifaunal benthos, which include mobile forms such as holothurians, gastropods, and scallops, tend to be a relatively minor biological component on soft bottoms. Attached epifauna on soft sediments are rare (see also *Important Species Interactions*) or involve highly specialized adaptations for attachment to fine-grained sediments (e.g., pennatulid coelenterates and some tunicates and hydroids).

Because soft bottom habitats are biologically complex, practicality dictates that biological sampling focus on those biotic components that are most easily sampled, counted, and identified. Macrofauna and those megafauna and demersal fish that are important to local fisheries are the most frequently used components for monitoring (U.S. EPA 1990). Estuaries are geologically transient features, so there are no species endemic or unique to a particular estuary. However, several commercially important biological resources exist in estuarine soft bottom habitats, including oysters, clams, mussels, crabs, lobsters, conchs, eels, and groundfish. Large and long-lived sedentary species (e.g., venerid bivalves) also serve as important indicator species because their presence indicates that the habitat is relatively stable and has sufficient quality to support their survival over several years. In those parts of the bottom that are infrequently reworked by wave or current energy and remain unpolluted, a diverse range of polychaetes, crustaceans, echinoderms, and molluscs dominate the infauna (e.g., Pearson and Rosenberg 1978). With increasing physical disturbance or pollution, the benthic



Note:
The progressive infaunalization following primary succession. A. Colonization of dredged material disposal mounds in Long Island Sound, USA. B. A pollution gradient away from pulp mill effluents in Sweden and Scotland.

Source: Reprinted with permission from Sigma Xi; Figure from Rhoads et al. (1978).

FIGURE 5B-2. Development of macrofaunal-sediment relationships over time/ space following either a physical or chemical disturbance.

TABLE 5B-2. BENTHIC ECOSYSTEM ATTRIBUTES ASSOCIATED WITH PIONEERING AND LATE STAGE SERIES

	Successional Stage	
System Attribute	Early (Stage I)	Late (Stage III)
Secondary production	High potential for r-selected taxa	High potential for K-selected taxa
Prey availability	High; prey are concentrated near surface	Low; infauna are deep burrowing[a]
Potential for food-web contamination	Highest for suspended or recently sedimented particulates; body burdens may be low related to short mean life spans	Highest for deeply buried contaminants; longer mean life spans may lead to significant body burdens
Contaminant/nutrient recycling	Limited to solutes in ≤3 cm	Solutes exchanged over distances to 20 cm or deeper
Potential for bottom-water hypoxia	High; storage systems for labile detritus	Low; a recycling or "purging" system

Source: Rhoads and Germano (1986)

[a] Nonlethal predation of distal ends of siphons or caudal segments may be important for some predator species.

system tends to revert to an assemblage of small, fast-growing, short-lived, opportunistic polychaetes/oligochaetes (r-selected species) (Figure 5B-1), or in extreme cases, to abiotic conditions.

Unlike open ocean food chains, which typically consist of up to five trophic levels (Ryther 1969), the estuarine food web has fewer levels (approximately three), with a greater diversity of inputs and alternative trophic pathways. These many trophic "options" impart relative stability to the estuarine food web. A generalized and simplified box model of an estuarine food web is shown in Figure 5B-3. The base of the food web structure of estuarine soft bottoms is largely based on phytodetritus in the form of vascular plants, seaweeds, and phytoplankton (Roman and Tenore 1984). Much of the detritus becomes finely divided over time and, in aerobic conditions, ultimately ends up as a relatively refractory organic fraction within the sediment column.

A more detailed food web is presented in Figure 5B-4, which shows the feeding relationships of detritus and grazing food chains. The source of the detritus is qualitatively different in different latitudes and salinity regimes as a result of changing species composition and relative proportion of detritus from algae, marsh grasses, sea grasses, and woody plants such as mangroves. For example, the food chain of subtropical to tropical lagoons and estuaries is very different for red mangrove (*Rhizophora mangle*; more saline conditions) than for black mangrove (*Avicennia nitida*; more brackish) (Figure 5B-5). Primary consumers of the suspended and deposited detrital pool include suspension-feeding and deposit-feeding invertebrates, respectively. Detritivorous fish (e.g., mullet) also use the detrital pool. Some invertebrates can exploit both suspended and deposited detritus (e.g., tellinacean bivalves, spionid polychaetes). Most macrofauna consume primarily either suspended or recently deposited fresh labile detritus or deeply buried more refractory detritus (Levinton 1972; Rice and Rhoads 1984).

Keystone species are species that exert a major influence on the presence, absence, abundance, or distribution of other species in the system. As such, keystone species are important to identify because, if they are lost from a system, the community may respond in a non-linear way. Examples of keystone species are herbivores that graze back weedy species, infauna that change bottom topography or sediment stability and therefore exclude or limit the distribution of other species (see *Key Ecological Processes* below), and high level predators that limit the population growth of certain prey species.

Key Ecological Processes

Nutrient Sources and Distribution—The nutrient sources for the base of the soft bottom habitat food chain are dissolved forms of nitrogen, phosphorus,

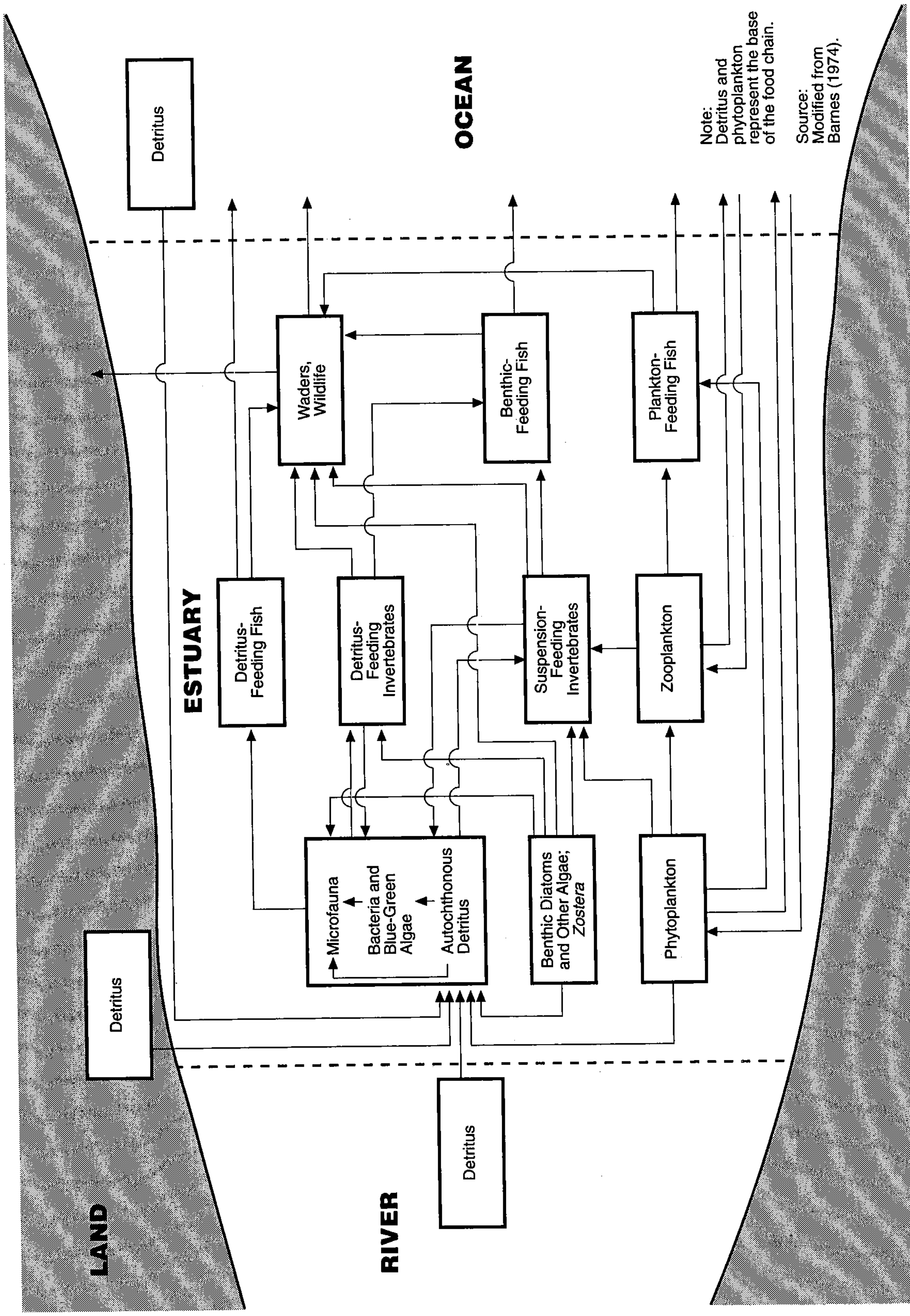


FIGURE 5B-3. A simplified generic estuarine food web.

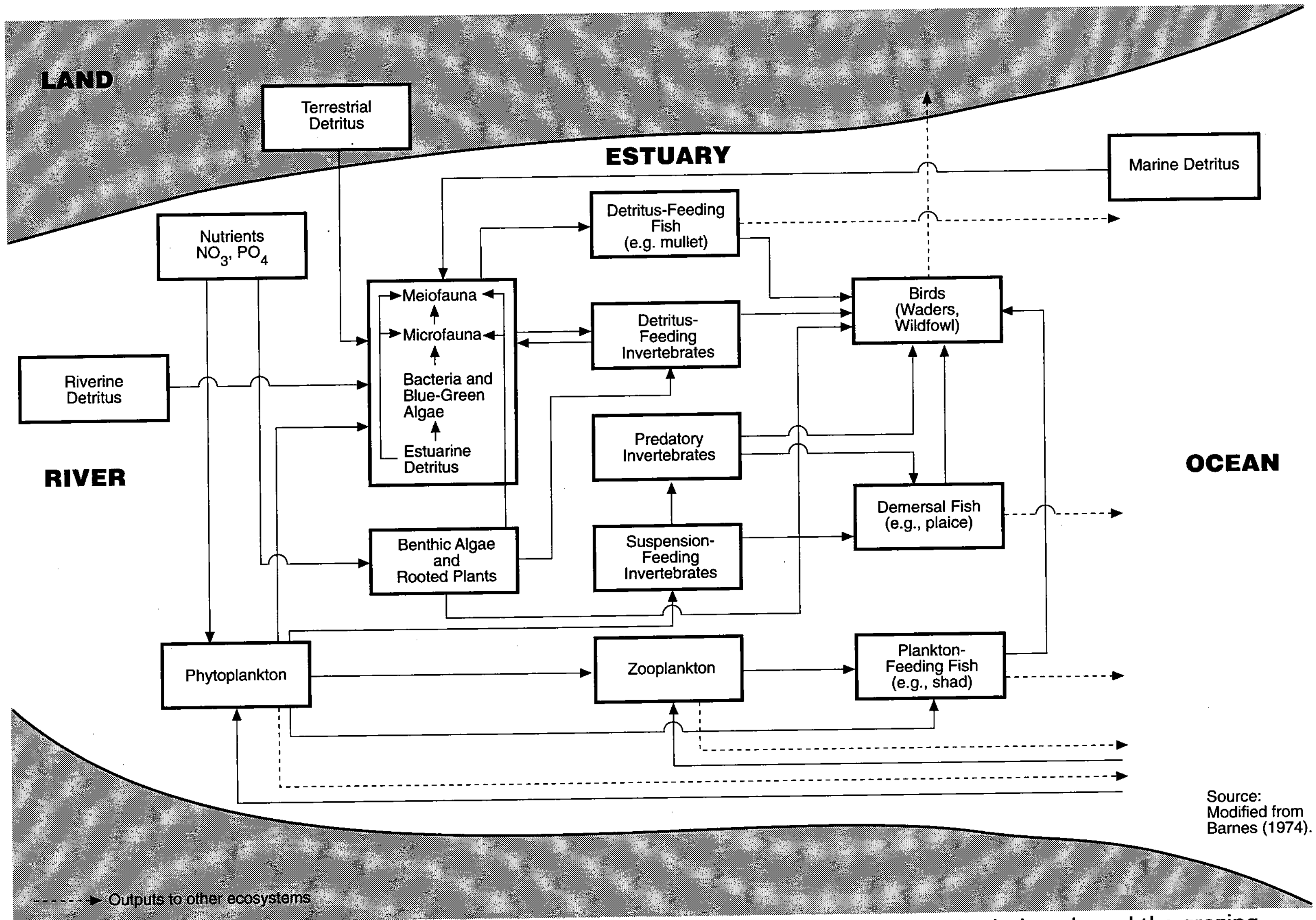
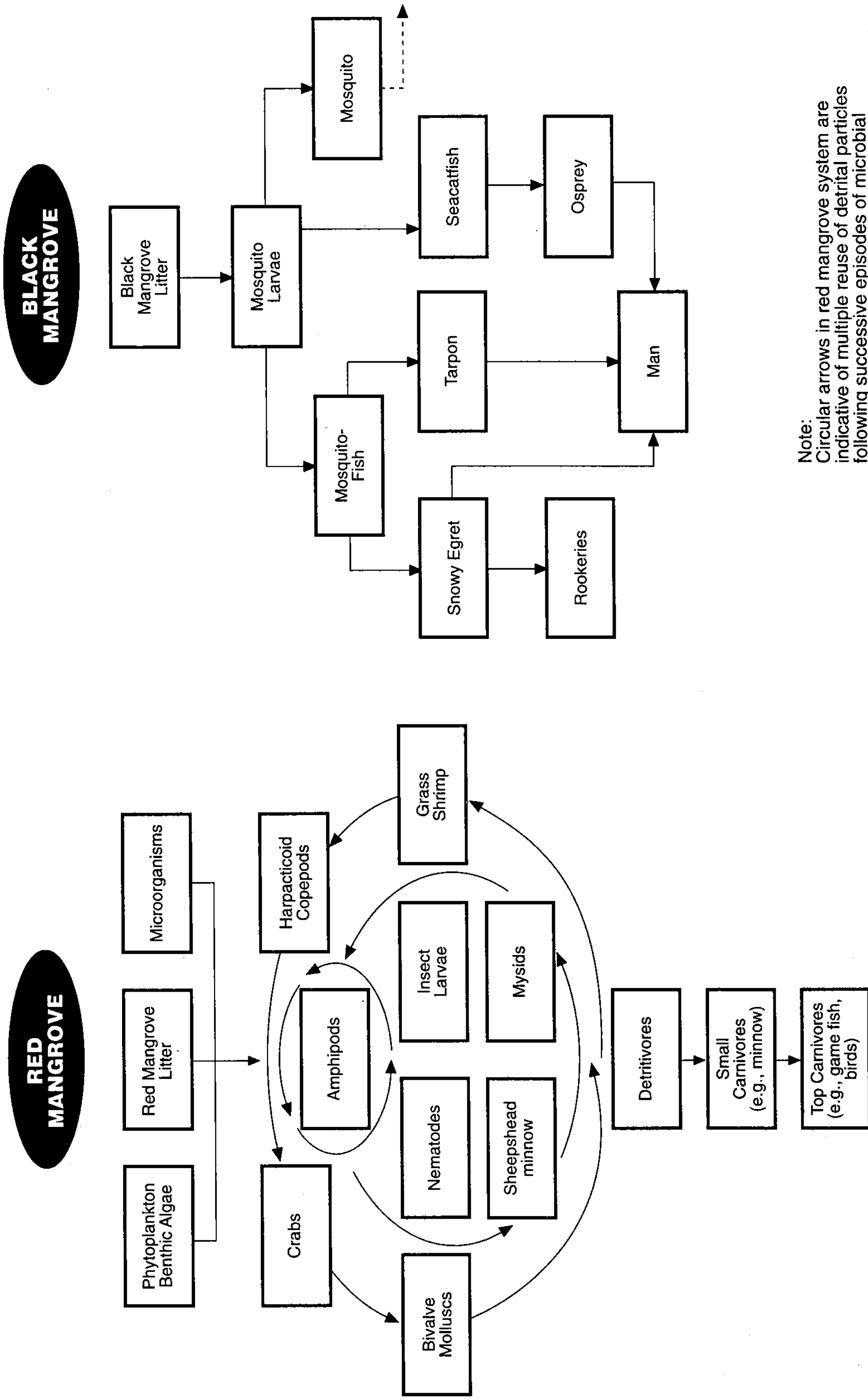


FIGURE 5B-4. A generalized food web showing the relationship between different detrital pools and the grazing food chain in an estuary.



Note:
Circular arrows in red mangrove system are indicative of multiple reuse of detrital particles following successive episodes of microbial colonization.

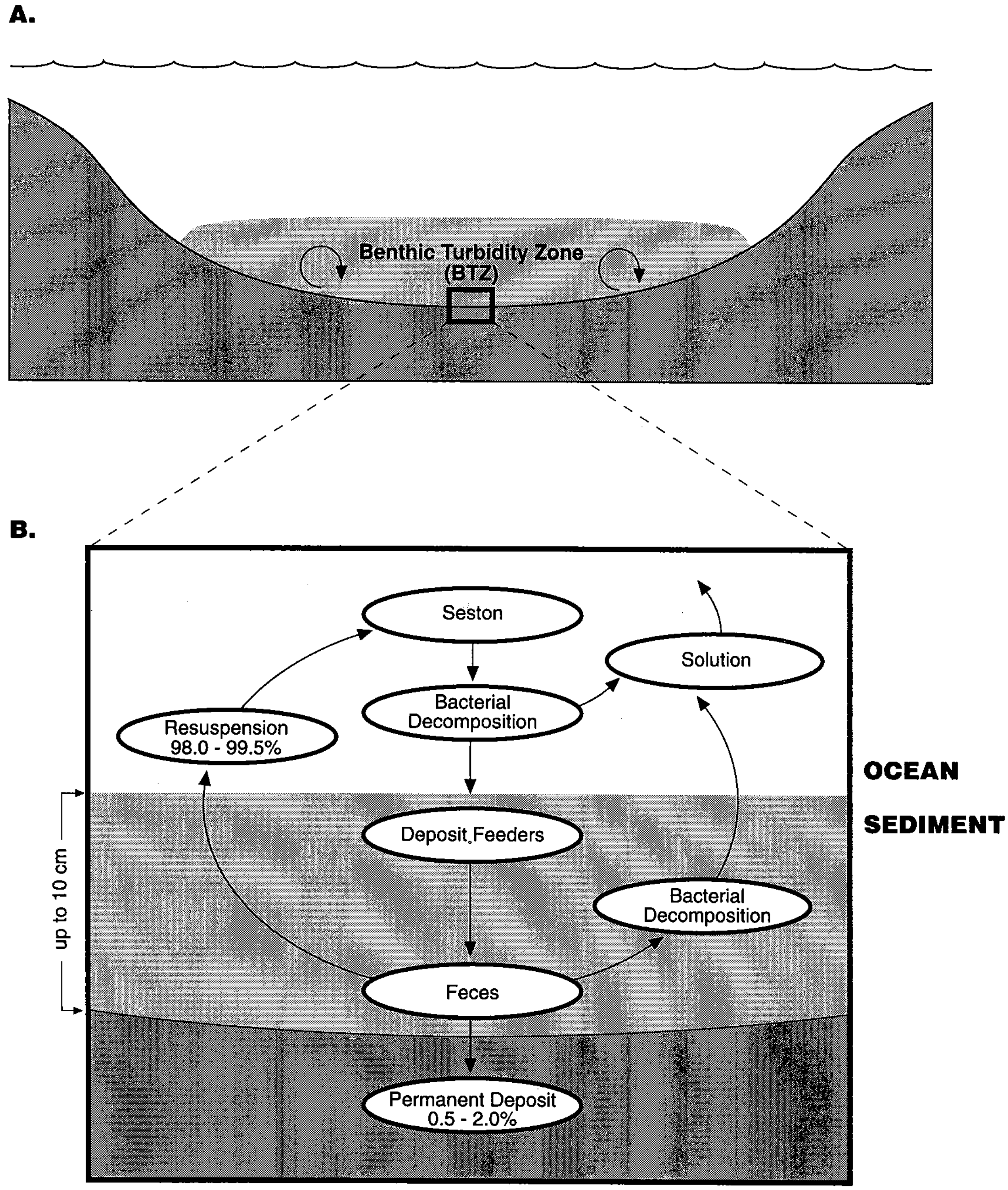
Source: Modified from Boaden and Seed (1985)

FIGURE 5B-5. Detrital food chains in Florida mangrove systems.

silicon, trace metals, and organic trace substances that include critical vitamins for plant growth (Kennish 1990). These nutrients enter estuaries from 1) river or stream inputs, nonpoint runoff, and groundwater; 2) the atmosphere in the form of precipitation or aerosols; 3) the open ocean; and 4) the bottom of the estuary itself (recycled). Nutrients for consumers are generated within the estuary and include living planktic species as well as benthic plants and particulate organic matter (POM) from decaying biomass. Organic detritus also enters an estuary in the form of riverine and terrestrial detritus, the open sea, and inputs from above (sea birds) (Kennish 1990).

In addition to inputs of "new" nutrients from the above sources, the estuarine nutrient cycle involves a great deal of internal recycling between dissolved organic matter (DOM) and POM. Phytoplankton and benthic plants fix carbon from carbon dioxide to form POM by converting soluble nutrients to biomass. The primary producer/primary consumer POM pool is then converted in the food web into other forms of biomass (microbial, nekton, benthos), and at each trophic level, DOM is excreted or lost by excretion or inefficient assimilation. Some of the unconsumed POM ultimately settles to the bottom and enters a sediment-water column recycling system (Figure 5B-6). More than 50 percent of the phytoplankton production and 90 percent of benthic macrophyte production passes to the detrital food web (Kennish 1990). The focusing point for much of this detritus is the soft bottom facies. The quantity and quality of POM actually ending up in the permanent deposit is a function of sedimentation rate, lability of the POM, and recycling efficiency at the sediment-water interface. Areas of soft bottom that tend to be eutrophic sinks for POM tend to have high sedimentation rates of fine-grained sediments and low dissolved oxygen and/or are in an early stage of succession (Forbes et al. 1994).

Important Species Interactions—The spatial-temporal disturbance history of soft bottom and trophic interactions are first-order factors for structuring species diversity, abundance, and biomass. The three variables of seasonal temperatures/photoperiod, disturbance, and grazing/predation interact to determine the dynamics of soft-bottom ecosystems. The timing and success of reproduction for consumers is linked to peaks in primary production cycles. In highly seasonal environments, this coupling may have significant time lags if phytoplankton peaks occur during low temperature periods (e.g., February) when benthic respiration is low. This is particularly true for estuaries in New England. The efficiency of coupling primary production peaks to consumer respiration is apparently lowest in northern latitudes because of the time lag between production and consumption compared to seasonally equitable environments. For this reason, subtropical to tropical estuaries may have higher assimilative capacities for organic matter than northern estuaries (see discussion in *Detrital Processing and Nutrient Regeneration*).



Note:
Intensive bioturbational mixing of bottom muds prepares the sediment for resuspension in the presence of relatively weak bottom currents. A. A benthic turbidity zone exists over many subtidal soft sediments related to tidal resuspension of the upper 2-3 mm of the bottom. B. Over 98% of the detritus settling to the seafloor is recycled and remineralized in the sediment column or overlying water column. This natural mixing and aeration process is analogus to tertiary sewage treatment. Loss of bioturbating organisms in stressed habitats can stop this process with the result that nutrient recycling and mineralization is arrested.

Source: Modified from Young (1971).

FIGURE 5B-6. Sediment-water column nutrient recycling.

Seafloor disturbance also contributes to stochastic cycles of secondary production. Pioneering stages of seafloor colonization following a disturbance tend to consist of low-diversity and high-density assemblages of rapidly growing opportunists. Biological assemblages are always recovering from past disturbances (Figure 5B-2). Disturbance results in enhanced diversity through the process of succession (Reice 1994), with the early stages of succession having enhanced secondary productivity (Rhoads et al. 1978). This high secondary productivity attracts predators such as fish and/or macrocrustaceans (Becker and Chew 1987). This predation pressure can, in turn, change the density of opportunists and rate of succession (Germano 1983).

Soft bottom, muddy habitats tend to be dominated by deposit-feeding species. Filter-feeding species, particularly epifaunal attached forms, are often rare or absent or are forced into patchy refugia (Rhoads and Young 1971). The paucity of this latter trophic group has been attributed to the phenomenon of trophic group amensalism (i.e., the exclusion of filter feeders by deposit feeders; Rhoads and Young 1970). Trophic group amensalism comes about by the intensive bioturbational activities, which destabilize muddy sediments, of soft bottom deposit feeders. In the presence of bottom currents, the loose, bioturbated surface can be resuspended by burying or otherwise clogging the filtering structures of suspension feeders. Trophic group amensalism is a natural process that limits both the numbers and kinds of species that can successfully colonize late successional stages on muddy bottoms. Species richness is typically depressed relative to physically stable subtidal bottoms. This observation is important to note from the perspective of habitat restoration. High order stages of succession may have a lower diversity than intermediate stages of succession (the "ecotonal" value). The absence of a species from a habitat may have more to do with competitive exclusion by a "keystone" species than it does to overall environmental quality. For example, the tube-dwelling amphipod *Ampelisca abdita* is characteristically absent from deep-water muds in Long Island Sound. This absence is not because of degraded habitat quality; rather, it is excluded by high sediment flux associated with intensive bioturbation. *A. abdita* is typically a transient ecotonal species excluded by late stage bioturbating species. Persistent populations of *A. abdita* tend to live on bottom types where bioturbation is not a sediment destabilizing factor. In this case, if the goal of habitat restoration is to optimize a habitat for *A. abdita*, one would need to periodically retrograde succession through anthropogenic substrate manipulation.

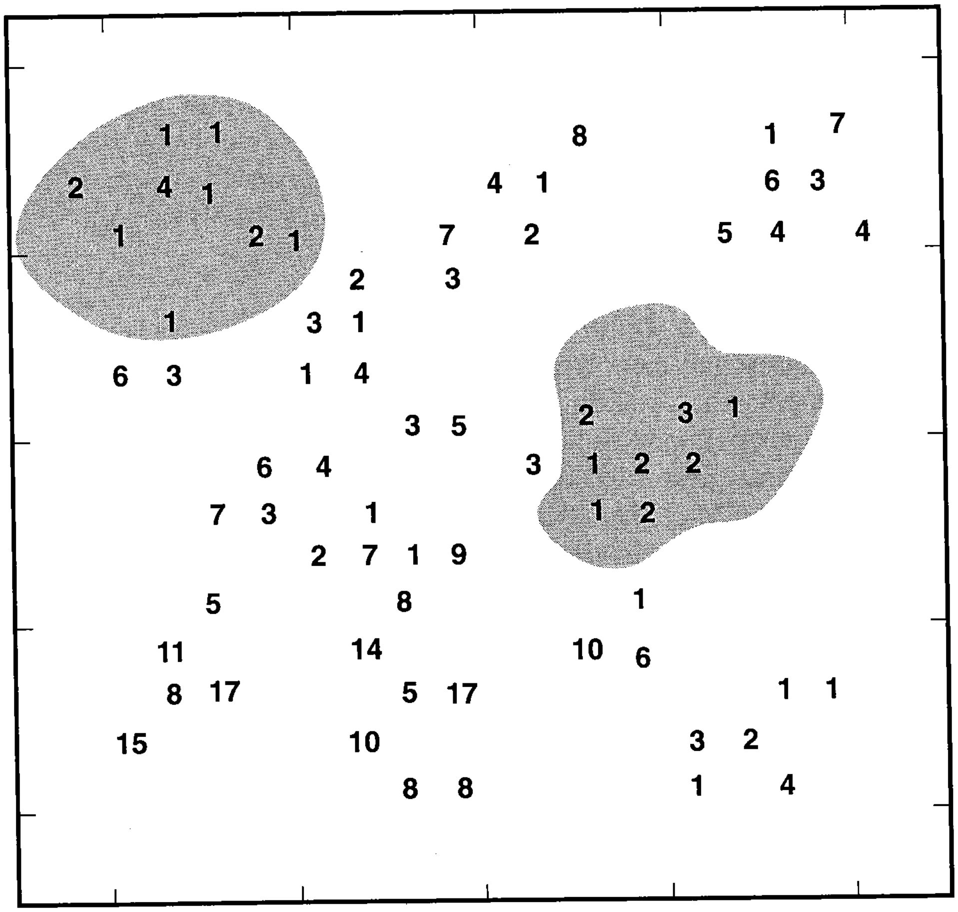
Detrital Processing and Nutrient Regeneration—POM settling onto a subtidal soft bottom rapidly enters the grazing food chain if water temperatures are sufficiently elevated to sustain high benthic metabolism. Estuaries located in climates where water temperatures seasonally range from more than 20°C to less than 10°C may be less efficient at nutrient recycling than those located in seasonally equitable areas. Detrital processing and nutrient regeneration in highly seasonal environments may only be effective for detrital processing in the spring, summer, and autumn, while detrital storage takes place during the winter (see also *Important Species Interactions* above). POM is represented by a wide range of substrates representing degrees of lability (availability to consumers). The most labile fractions are fresh phytoplankton, zooplankton, and carcasses of fish and benthic invertebrates and the bacteria associated with this decaying material. The next most labile fraction is seaweed (a species complex that has its own order of lability [Tenore et al. 1984]). The leaves and stems of submerged vascular plants are the next most labile fraction. The most refractory component consists of leaves and stems of marsh grasses and highly polymerized terrestrial carbon (Tenore et al. 1984). The most labile fractions are rapidly recycled near the sediment-water interface (Martens 1984), but the more refractory organic matter requires extended microbial decomposition (Valiela et al. 1984). Decomposition rates of relatively refractory organic matter can be significantly accelerated by bioturbation (Yingst and Rhoads 1980), particularly by head-down deposit feeders (Rice and Rhoads 1984). The regeneration of nutrients back into the water column from sediments is most efficient in those parts of the sea floor where deep bioturbation takes place, particularly during periods when elevated water temperatures result in high benthic metabolism (Aller 1982).

The goal of a soft bottom habitat restoration project may be to reverse deleterious effects of organic enrichment (eutrophication). For this to be effective, the input rate of labile organic matter must be reduced to match the ability of the system to aerobically respire POM and DOM. A review of the few literature sources available on organic loading rates in seasonal coastal environments suggests that loading rates of up to approximately 500 g carbon/m^2-year results in efficient aerobic recycling of organic matter without creation of a large pool of oxygen consuming metabolites such as hydrogen sulfide, methane, or ammonia (Valente et al. 1992). Loading rates greater than this critical threshold appear to lead to impaired aerobic recycling and the net loss of dissolved oxygen due to high biochemical and chemical oxygen demand. In those environments where water temperatures tend to remain high all year long (subtropical to tropical), the critical organic loading threshold may be higher than that stated above because of higher microbial decomposition rates on a yearly basis. This assertion is not quantified for marine sediments but is supported by analogous north-south gradients in terrestrial soil types: humic-rich soils in warm/cold climates as contrasted to humic poor "red" soils in warm and equitable climates.

Habitat Heterogeneity—The subtidal soft bottom environment tends to be relatively homogeneous compared to intertidal and wetland environments. Nevertheless, level soft bottoms can exhibit patchiness related to gradients in grain size and composition, reflecting different sediment sources, sedimentation rates, and/or levels of kinetic energy. Patchiness is commonly related to depth and correlated to the frequency and intensity of wave reworking or gradients in trawling activity. Chemical patchiness results from gradients in natural or anthropogenic deposition of organic matter and/or contaminants. Heterogeneity in water quality can be caused by local hypoxia and ecological (successional) mosaics related to patchy disturbances (Johnson 1972) (Figure 5B-7).

Key Natural Disturbances—The five most important systemwide natural disturbances that can potentially affect soft bottom habitats are 1) foraging by large predators, 2) storm turbulence, 3) quantum inputs of new sediment, 4) massive erosion, and 5) regional hypoxia. In high latitudes, benthic foraging by the California gray whale and Pacific walrus in the Bering Sea produces pits and furrows on a scale that rivals disturbances by geological processes (Nelson and Johnson 1987). Equivalent benthic disturbances in more southerly latitudes can be produced by foraging rays. Soft bottom sediments normally accumulate in low kinetic energy areas of the estuarine seafloor that are protected from wave or current resuspension. During extraordinary storm events, resuspension of soft sediments can take place, dispersing both sediments and infauna. These high energy events can also introduce coarse-grained sediment from nearshore into otherwise soft bottom habitats, temporarily changing sediment type. "Wet" hurricanes or major rainstorms within a drainage basin can promote unusually high sedimentation rates during small time periods (Scott et al. 1969; Hayes 1978). Hurricanes, while infrequent, can cause major changes in estuarine shoreline configuration, bathymetry, sediment distribution, sedimentation rate, and biological facies (Hayes 1978). Quantum sedimentation/erosion events result in redistribution, burial, and suffocation of infaunal communities. The critical burial depth for killing soft bottom benthic species by instantaneous burial ranges from less than 5 cm to more than 50 cm (e.g., Kranz 1974).

While lowered dissolved oxygen can be caused or exacerbated by anthropogenic activities, natural hypoxia can develop regionally within an estuary as the result of naturally high runoff events that cause salinity stratification that can inhibit advective transfer of atmospheric oxygen through the water column to the bottom. Dissolved oxygen values less than 2 mL/L induce behavioral avoidance of low oxygen areas, while values less than or equal to 1.0 mL/L (hypoxia) cause physical inactivity and, under prolonged periods, mass mortalities (Tyson and Pearson 1991). Hypoxia accompanied by mass mortalities can result in accumulation of anaerobic metabolites such as hydrogen sulfide and/or methane gas and ammonia in the affected bottom. With re-aeration of the soft bottom habitat, recolonization may take place by immigration of mobile adults and/or larval recruitment. Restoration projects should consider the potential effects of all of



Note:
This diagram represents a few m^2 of seafloor (scale undefined). Individual organisms are shown as numerals. The value of the numeral indicates the rank of the species in the order of succession (see Figure 5B-2). The shaded areas are those most recently disturbed.

Source: Modified from Johnson (1972).

FIGURE 5B-7. Biological patchiness in an otherwise homogeneous physical/chemical environment reflects past disturbance events.

these natural processes on the restoration process or structure(s) because the long-term success of a low maintenance project will be determined by these natural disturbances.

Landscape Interactions—The watershed adjacent to a normal estuary can be a major source of both dissolved and particulate nutrients and contaminants. Most of these materials are introduced by sewer outfalls, roadway or agricultural drainage pipes/canals, surface runoff from streams and rivers, or groundwater discharge. Some nitrogen and contaminants can be introduced as aerosols or by precipitation. Nutrient and contaminant balance calculations for an estuary are incomplete unless all of these potential sources are quantified. Once introduced into an estuary, soluble nutrients are rapidly converted into biomass. Organic and inorganic particulates and contaminants adsorbed onto particles ultimately end up on the bottom. Over time, the finest-grained and most organic-rich particles are redistributed by waves and currents until they settle into the lowest kinetic energy areas of an estuary. Commonly, this site is the subtidal soft bottom habitat below the mean storm wave base.

Lagoonal estuaries, which have reversed estuarine flow relative to normal estuaries, often have ambient turbidity levels less than those of normal estuaries because land-derived runoff from the adjacent watershed is, on average, less (Postma 1969). During brief seasonal periods of runoff, the watershed becomes a more important factor than the ocean for introduction of nutrients, sediments, and contaminants.

Functional Values

The functional value of the subtidal soft bottom habitat to overall ecosystem function is biomass production and organic matter/nutrient recycling. In well-illuminated, shallow areas, primary production is locally important in the form of seagrasses and benthic diatoms. This production is both a source of consumer food and an important refugium for juvenile stages of fish and scallops. Dense stands of SAV, such as *Zostera marina*, *Thalassia testudinum*, and shoalgrass (*Halodule wrightii*) also serve to bind and stabilize sediments (see also *Estuarine and Coastal Wetlands*). Unvegetated soft bottom habitats also are productive in terms of infaunal detritivores and predators, which form the base of a consumer food chain that leads to many commercially important species such as crabs, lobsters, flatfish, skates, conchs, and cod.

Biological activity in soft bottom environments serves to recycle nutrients back into the water column from decaying detritus (Aller 1982). This recycling is enhanced by particle and pore water bioturbation. The bioadvection of pore water into and out of the bottom serves to supply oxygen and sulfate to the sediment column for efficient microbial mineralization. Advection of ammonium and silica out of the bottom into the overlying water column can be a significant nutrient feedback to estuarine phytoplankton (Aller 1982). An analogy can be drawn between bioturbation and tertiary sewage treatment, as they both involve particle stirring and water aeration. The importance of bioturbation for oxidative metabolism of organic matter and nutrient recycling depends on the successional status of the infaunal community. High-order successional stages tend to be more effective in deep bioadvection than pioneering stages (Rhoads and Germano 1986; see also Figure 5B-2 and Table 5B-2).

Causes for Deterioration

A major cause of subtidal soft bottom habitat deterioration is the loss of SAV as a result of disease and epiphytic growth on leaves. This loss results in an overall decrease in primary benthic production because plants are buried by sedimentation/erosion and shaded by suspended seston. Organic enrichment and/or dredged material, which can compromise benthic recycling as described above, can also cause habitat deterioration. Sediment contamination by metals, fuels, pyrogens, polychlorinated biphenyls, and pesticides may degrade habitat value by causing mortality, or the exposed species may experience impaired behavior, physiology, morphological defects, biochemical abnormalities, impaired reproduction, and alterations in community structure and function (Sheehan 1984). Even if secondary and higher level production is maintained, contamination of the food chain imparts a negative value to a habitat from a fisheries perspective. Regional hypoxia/anoxia is another factor that can compromise secondary production by eliminating key species in the benthic food chain.

Assessment of Habitat Health

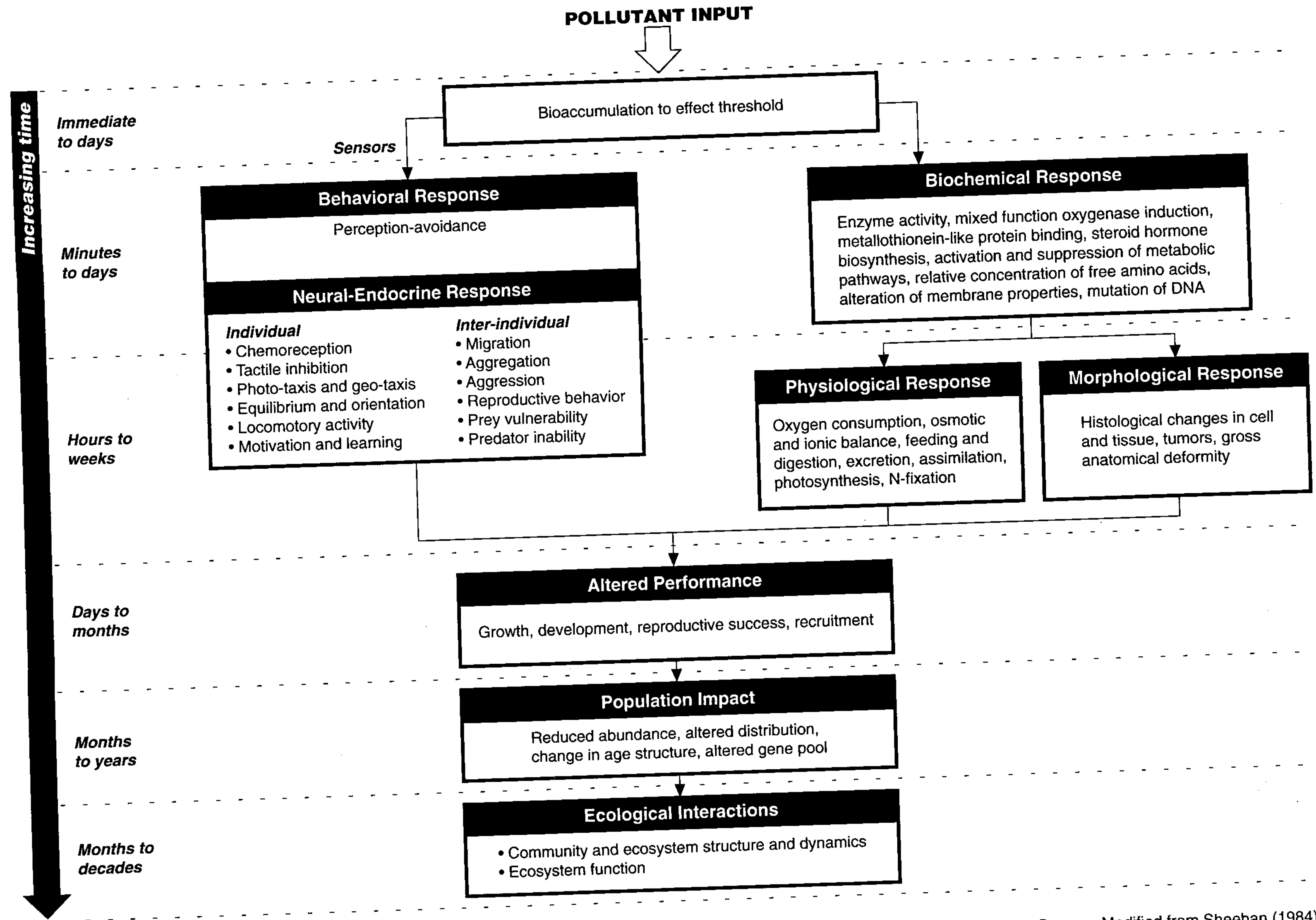
The concept of habitat health presumes that the habitat of interest has a potential for realizing some high level of functionality. From an anthropogenic view, a negative change in production, diversity, or recycling ability is a manifestation of compromised health. In habitat restoration, it is important to realize that some habitats, even in the absence of anthropogenic influence, may not be capable of functioning well, even under pristine conditions. For example, parts of estuaries may have experienced high organic and high metal loading, bottom anoxia, and azoic conditions long before the appearance of man (e.g., Brumsack 1991). Therefore, environmental health is a relative concept, and it may be impractical, for example, to attempt to restore high functionality to a habitat if it has a natural tendency to be hypoxic. The overall functions of a particular habitat can be compared relative to historical baseline conditions or to an adjacent reference habitat.

Figure 5B-8 shows the chronology of effects that can cascade through an ecosystem when pollutants are introduced. These effects relate both water column and benthic species, but benthic macrofaunal species tend to be the focus as sentinels of system health. The reason for a benthic emphasis is that benthos tend to be more sedentary and have longer mean life spans than most planktic species. These attributes make the macrobenthos and demersal fish better time-integrators of overlying water quality than organisms actually living within the water column (plankton and nekton). The potential list of measures of ecosystem health is large, but practicality usually dictates that only a few be used. A critique of indicators is provided in Scott (1989). The U.S. Environmental Protection Agency's (EPA) Environmental Monitoring and Assessment Program has recommended 12 parameters for monitoring ecosystem health and includes guidance on how each indicator is to be applied to assessment, period of measurement, recommended methods, expected range of variability, problems of measurement and/or interpretation, and key references (U.S. EPA 1990). The 12 parameters are: 1) dissolved oxygen, 2) benthic abundance, biomass and species composition, 3) biological sediment mixing depth, 4) extent and density of SAV, 5) fish abundance and species composition, 6) presence of large indigenous bivalves, 7) fish pathology, 8) acute sediment toxicity, 9) chemical contaminants in sediments, 10) water clarity, 11) water column toxicity, and 12) chemical contaminants in fish and shellfish. A particular habitat restoration program may not require all 12 parameters to be measured to assess habitat status. The nature of the restoration project will dictate the kinds of questions to be asked and the appropriate indicators to be measured. Establishing these questions and developing a tiered monitoring approach is critical for developing an efficient and successful restoration project (NRC 1990).

Hard Bottom Habitats

Naturally occurring hard bottoms are generally rare in estuarine habitats because of the persistent sedimentation that accompanies riverine influences. Most often, natural hard substrate is limited to rock outcrops or coarse glacial deposits elevated above the surrounding sandy and muddy level bottom, remnant glacial-carved walls, or cobble/gravel beaches in relatively shallow, high energy environments. These are more frequently found in mesohaline (salinity 5–18 ppt) or euhaline (salinity 30–40 ppt) reaches of an estuary, farther away from the predominant sources of siltation accompanying runoff and erosion in the fresh-water and oligohaline reaches.

There are numerous examples of anthropogenic hard bottom structures that have been introduced into estuaries as either habitat (e.g., artificial reefs, shell bottom, gravel beds), inlet and channel stabilization structures (e.g., rock jetties), or over- or underwater construction support (e.g., pilings, retaining walls). These



Source: Modified from Sheehan (1984).

FIGURE 5B-8. A conceptual chronology of effects following exposure to toxic pollutants.

structures provide refuge, food, and spawning habitat for a variety of invertebrate, fish, and bird species. The successes and failures of these various anthropogenic structures in creating vertical habitat in the estuary are relevant to estuarine restoration discussions.

Three types of hard bottom habitats are discussed below: rocky shores and gravel/cobble beaches (natural substrates), reefs and in-bay terraces (artificial substrates), and modified substrates.

Natural Substrates: Rocky Shores and Gravel/Cobble Beaches

Geographic Distribution—Naturally occurring rock substrate is more typically found in the glacially formed inlets and fjords of the Atlantic Northeast and Pacific Northwest, including Alaska. Estuaries in the mid to south Atlantic, the Gulf of Mexico, and southern California tend to be dominated by soft-bottoms.

Zonation Within Habitats—Hard bottom community structure varies from those typical of open ocean systems to those found in brackish marshes. For example, subtidal rock within Elliott Bay in Puget Sound, Washington, supports growth of *Nereocystis* spp. and *Laminaria saccharina*, kelps that are also found in protected coastal waters from Point Conception in California to Alaska. The algae *Fucus distichus* has been documented to occur on rocks and logs high up in the Squamish River delta of Howe Sound in British Columbia, Canada, and to be mixed within the sedge *Carex lyngbyei* (Thompson 1982). The algae *Fucus serrata* is also commonly observed in the Atlantic Northeast estuaries. Oyster and mussel distribution can occur relatively high up into the lower mesohaline reaches of estuaries on all coasts; residence is typically governed by the length of time of freshwater intrusion.

The intertidal and subtidal zonation discussed in Figure 5A-1 is commonly observed in rocky polyhaline and euhaline reaches of the estuary. Common algal assemblages include *Enteromorpha* spp. and *Ulva* spp. in the upper intertidal and *Fucus* spp. and/or *Ascophyllum* spp. in the lower intertidal, grading into kelp communities in the subtidal. Animal communities are also segregated by tidal level, although salinity gradients play an important role in estuaries. Typically, solitary sessile organisms, animals that possess hard external body coverings (e.g., periwinkles, barnacles, mussels, limpets), dominate the intertidal zone. Subtidally, soft-bodied invertebrates (e.g., *Metridium* spp.) and colonial animals dominate hard substrate (e.g., corals, bryozoans, sponges). Motile invertebrates also use rock substrate for foraging or shelter (e.g., amphipods, crabs, shrimp, lobsters).

In mesohaline or oligohaline (salinity 0.5-5.0 ppt) environments, overall diversity tends to decrease depending on the freshwater intrusions or saltwater wedges that form and collapse during changes in river output. It is not uncommon to have a freshwater lens form on top of deeper saltwater, with accompanying biota forming in each habitat. For example, surface waters on the Neches River above Sabine Lake at Beaumont, Texas, have been measured at 0–2 ppt, while the bottom waters have been measured at 11 ppt (SAIC 1994).

Biological Community—The species, food web structures, and community dynamics within naturally occurring hard bottoms tend to be site-specific and are dependent on a combination of physiological factors (e.g., geographic location, salinity, temperature) at the site. However, similar species tend to occupy similar niches for all coastal estuaries, as noted in Table 5B-3.

Estuaries are highly productive regions for a large variety of commercially and recreationally important fish and shellfish species. These species can range from oysters, mussels, crabs, or lobsters, which use rock surfaces for substrate or shelter, to recreational or commercial fish species that use kelp canopies as a source of both food and shelter. Pacific herring (*Clupea harengus pallasi*) spawn in estuaries on kelp blades or on eelgrass. Migrating juvenile salmonids forage actively in cobble and gravel for epibenthic organisms (e.g., *Anisogammarus* spp., *Crangon* spp., amphipods [*Corophium* spp.]).

Naturally occurring rock habitat is also an important resource for avian species. While often not strictly estuarine, numerous permanent and migratory species use the plants, invertebrates, and fish in estuaries as food sources. For example, black brant (*Branta bernicula*) actively feed on algae (*Ulva* spp. and *Enteromorpha* spp.) and Pacific herring roe attached to kelp, when available. Bald eagles (*Haliaeetus leucocephalus*), double-crested cormorants (*Phalacrocorax aurtius*), and osprey (*Pandion haliaetus*) forage for juvenile salmonids over cobble or in kelp.

Mammals are also important users and consumers in estuarine hard bottom communities. Harbor seals (*Phoca vitulus*) and California sea lions (*Zalophus fornianus*) are year-round residents of Pacific coast estuaries, foraging on fish and shellfish on both soft and hard bottom habitats. An often overlooked, but important, life-function for both mammals, is the use of exposed rocks as haul-outs for sunning, calving, or escape from predators. Intertidal rock or gravel/cobble habitats are used by terrestrial mammals for foraging. Raccoons (*Procyon*

TABLE 5B-3. EXAMPLES OF WEST AND EAST COAST HARD BOTTOM NICHE SUBSTITUTIONS, GULF COAST EQUIVALENTS, AND TROPICAL TYPES WHEN STRESSED

System Type	Description of Role	Tropical Stressed	Upper West Coast	Gulf Coast	Upper East Coast
Oligohaline river mouth	Oyster reef niche	American oyster (*Crassostrea virginica*)	Pacific oyster (*Ostrea gigas*)	American oyster (*Crassostrea virginica*)	American oyster (*Crassostrea virginica*)
Middle salinity estuary	General crab carnivore, moving into and out of varying salinity	Blue crab (*Callinectes*)	Dungeness crab (*Cancer magister*)	Blue crab (*Callinectes*)	Green crab (*Cancer*)
High salinity estuary	Top carnivore in bottom irregularities	Spiny lobster (*Panulirus*)	King crab (*Paralithodes*)	Stone crab (*Menippe*)	American lobster (*Homarus*)
Kelp system	Algal forests, bottom attached seaward of surf		*Macrocystis/ Nereocystis*		*Laminaria*
Intertidal rock	Grazers of intertidal rocks, periwinkles	*Littorina ziczac*	*Littorina planaxis* *Littorina scutellata*	*Littorina irrorata*	*Littorina litorea*
	Intertidally protected filter feeders, barnacles	*Balanus amphitrite*	*Balanus glandula*	*Balanus eburneus*	*Balanus balanoides*
	Sea lettuce	*Ulva*	*Ulva*	*Ulva*	*Ulva*
	Rockweed		*Fucus distichus*		*Fucus spiralus*

Source: Odum et al. (1974)

lotor) and river otters (*Lutra canadensis*) feed on crabs, fish, or shellfish found in quieter intertidal areas of the estuary.

Key Ecological Processes—The same physiological factors that affect soft bottom estuarine communities also influence the types of habitats that can form on hard substrate. Salinity, substrate (rock vs. cobble), and tidal height are perhaps the dominant physiological factors determining community structure, but light, temperature, depth, suspended solids (turbidity), currents, and water quality all combine and influence the type of community that can form within a specific estuarine gradient. Biological factors that influence habitat structure include competition for space or light and grazing or predation pressures.

Temperature and salinity gradients in estuaries of North America range from very stable environs with narrow ranges to extreme variability. Estuaries on the Pacific and northeastern Atlantic coasts form relatively stable gradients between river and sea, with little annual fluctuation in temperature and salinity. This is typical of estuaries such as Narragansett Bay, Rhode Island; Chesapeake Bay, Maryland and Virginia; Puget Sound, Washington; Prince William Sound, Alaska; and San Diego Bay, California. In contrast, Texas estuaries are characterized by year-round high temperatures, with low but extremely variable salinities (Lynch et al. 1976). In the South Atlantic Bight, which includes the Atlantic coastal states of North and South Carolina, Georgia, and Florida, large annual changes in temperature have major effects on the benthic community structure (Hay and Sutherland 1988).

When light is sufficient and all else is equal, subtidal colonial or sessile invertebrates lose in competition with plants. For this reason, seaweeds generally dominate in shallow sunlit habitats, while colonial or sessile animals dominate in darker subtidal or shaded habitats. Notable exceptions are the symbiotic anenomes (e.g., *Anthopleura* spp.), which co-exist in the algal zones by using the photosynthetic abilities of their symbiotic blue-green algae for food production.

While physiological factors control the upper elevation limit of estuarine communities, biological factors appear to control the lower elevation limits. For example, in high-energy environments, the lower distribution limits of barnacles are controlled by competition with mussels, while the lower limits of the mussel beds are affected by predations from starfish, sheepshead, and oyster drills (Hay and Sutherland 1988). In the South Atlantic Bight, the abrupt end to the oyster zone at mean low water is due to several biological processes, including drills and boring sponges. Oysters are found subtidally only where salinity excludes the other predatory species (Hay and Sutherland 1988).

In planning restoration efforts, it is fairly straightforward and common to measure the physiological processes needed for the target organisms and predict success or failure based on those parameters only. However, a complete understanding of biological factors is often overlooked or underestimated. Merrill and Gillingham (1991) successfully created a *Macrocystis pyrifera* kelp forest on long-lines in central Puget Sound where extensive stands of *Nereocystis* spp. naturally exist, but *Macrocystis* spp. does not. While physiological conditions were adequate to support adult sporophytes, the program was only successful by culturing gametophytes on culture strings in a laboratory and then, after sexual fertilization and sporophyte generation, planting the string in Puget Sound. The inference is that biological factors prevent or preclude *Macrocystis* spp. from completing its sexual life cycle in Puget Sound (Merrill 1995, pers. comm.). In another example, in cage enclosure experiments in the South Atlantic Bight, polychaetes and amphipods excluded barnacles and leafy algae in the upper intertidal. In nature, competition between these groups probably occurs all the time, but because amphipods and polychaetes are fish prey items, their abundance is usually controlled (Hay and Sutherland 1988).

Nutrient Sources and Distribution: By virtue of upland runoff, estuarine waters and sediments typically contain high concentrations of both macro- (e.g., nitrogen, phosphorus, potassium) and micronutrients (e.g., copper, zinc) that support both pelagic and benthic micro and macroalgae at the base of the food chain. Northern estuaries tend to be influenced more by tides that drive the currents that mix and distribute the nutrients and food chain resources. By contrast, Gulf of Mexico estuaries tend to be less tidally influenced and rely on river currents and wind-driven processes to distribute both abiotic and biotic nutrient sources.

Detrital Processing and Nutrient Regeneration: The rich algal productivity of estuarine rocky habitats augments the base of the food chain established by water-column phytoplankton blooms. Countless invertebrates either feed directly on the growing algal thallus or find a rich food source in the unattached algal detritus that finds its way to both hard and soft bottom habitats. Duggins et al. (1989) showed that more than half the carbon of some fish and birds in a kelp-dominated habitat can be traced to carbon ultimately fixed photosynthetically by kelp. There, invertebrates then serve as prey for other invertebrates, fish, birds, and mammals.

Habitat Heterogeneity: Estuaries are naturally heterogenous, and all the physiological factors discussed previously can vary within an estuary. Salinity in estuaries varies both in proximity to river or ocean, but also by depth.

Substrate can vary by slope, exposure, or cobble (i.e., solid vs. gravel). Light varies not only with depth, but also with seasonal suspended sediment loads. Water quality may change with proximity to sources of chemicals of concern or with seasonal high water outflow from rivers. At best, homogeneity of habitat can be achieved only in relatively limited areas.

Key Natural Disturbances: A key natural disturbance within all estuaries is freshwater input. The amounts and timing of freshwater input are both important. In northern climates, spring rains and melting snows combine to create large volumes of freshwater that occur for several weeks and alter estuarine salinity regimes. Strong tidal currents can quickly restore the saltwater/freshwater balance. By contrast, in southern regions, tropical storms will produce large floods in relatively short periods of time. Small tidal ranges result in reduced tidal mixing and lead to long periods of time where salinities remain low. Marine species not adapted to these long periods of reduced salinity will be eliminated by these conditions.

Landscape Interactions: Estuaries are affected by activities that occur in their watershed(s). Natural or anthropogenic upstream activities can have significant impacts within the estuary for both hard and soft bottom communities. Increased sedimentation due to logging, changes in sediment budgets, water quality and flow characteristics from dams or channelization, fertilizer and pesticide runoff from agriculture, and discharge of chemicals from industrial and municipal sewage outfalls all have direct effects on estuarine processes. Non-point sources of pollution from communities surrounding estuaries also have a direct effect on overall water quality and may impact biological resources.

Functional Values—As often stated, estuaries are among the most productive habitats within North America, supporting a diverse number of invertebrates and fishes, which in turn, sustain birds and mammals as well as recreational and commercial fisheries.

Causes for Deterioration—Changes in any physical or chemical environmental conditions that define the physiological boundaries within which populations exist represent changes in estuarine environments. Alterations in salinity resulting from river channeling or diking alter the flow and distribution of freshwater in the estuary and redefine estuarine salinity zones, consequently impacting biological systems. Conversely, the construction of navigation ship channels within the lower reaches of estuaries can result in the penetration of high-salinity waters into otherwise fresh- to brackish-water environments.

The effects of altering flows and salinity in estuaries can affect changes that are indirect and difficult to predict but nonetheless very significant. An excellent example exists at the mouth of the Mississippi River in Louisiana. Historical riverside levee construction, channelization, and other construction activities were designed to improve hydraulic efficiency and to achieve the benefits of flood control that resulted from moving floodwaters through the Lower Mississippi Valley and into the Gulf of Mexico more rapidly. With these higher flows, sediments that were previously deposited in coastal marshes along the lower river were flushed into the Gulf, thereby contributing to the apparent subsidence of Louisiana's coastal wetlands.

Turbidity/sedimentation has increased as a result of increased erosion from upland construction and logging efforts in the Pacific Northwest, causing dramatic effects on anadromous fish populations. On the other hand, dams constructed along those same rivers where logging has occurred have reduced overall flow into the estuaries, reduced sediment transport past the dams, and restricted access to upstream spawning grounds. Finally, the effects of chemicals associated with human activities, both from point and non-point sources are important, as discussed in other ecosystem sections within the *Ecosystem and Restoration Profiles* chapter.

Assessment of Habitat Health—There is generally no single indicator or sentinel of overall habitat health for rocky estuarine environments. Habitat Evaluation Procedures (HEPs) developed by the U.S. Fish and Wildlife Service can and have been applied to single estuarine species such as shrimp (Mueller and Whitehead 1986), great blue heron (USFWS 1985), and numerous others. While well documented, HEPs are basically predictive models that are influenced by the validity of the assumptions used to construct the models. HEPs have also been criticized because they focus on single species and do not deal well with important biological interactions such as competition and predation.

Another broader approach, the *Estuarine Habitat Assessment Protocol* (Simenstad et al. 1991), has been developed for Puget Sound, Washington, and can be used as a model for other estuaries in North America. This procedure stresses function in preference to simply assessing fish and wildlife use. A habitat is evaluated for its contributions to species needs for reproduction, feeding, refuge, and physiology. The authors' underlying assumption in the protocol is that identifying the characteristics of the habitat that *explicitly* promotes fish and wildlife are more important than simply identifying or quantifying specific-species usage. The protocol was developed with a specific aim of being able to compare post-restoration efforts with pre-existing conditions.

Artificial Substrate

Artificial habitats have been constructed in near coastal waters, bays, and estuaries throughout North America primarily as fish aggregating devices (Buckley 1989) but also as substrate for establishing or restoring kelp forests (MEC 1993). Artificial reefs and in-bay terraces are discussed in the sections below.

Reefs—Artificial reefs (though constructed over soft bottoms) provide a 3-dimensional habitat within the water column and likely represent the best opportunity for hard bottom restoration efforts in estuaries. This section briefly discusses the status of artificial reefs within estuaries as they relate to restoration, or replacement, of hard bottom communities. Various construction materials have been used, ranging from recycled materials (e.g., tires, fly-ash composites) to fiber reinforced plastic, but most commonly include quarry rock. Excellent reviews of state-of-the-art techniques can be found in Duedall and Champ (1991), Seaman and Sprague (1991), and in a dedicated series of articles on artificial reefs from the Fourth International Conference on Artificial Habitats for Fisheries (Seaman et al. 1989).

Geographic Distribution: More than 250 artificial reefs have been established in near coastal waters and estuaries of the United States, with the highest density in the South Atlantic and Gulf coasts (McGurrin et al. 1989; NRC 1994). Artificial reef habitats have been constructed in Delaware Bay (Sheehy and Vik 1992); Chesapeake Bay (Feigenbaum et al. 1989; Lucy and Barr 1989); San Diego Bay, California (Ambrose 1994; MEC 1993); Florida (Davis 1985); and Puget Sound, Washington (Hueckel et al. 1989).

Zonation Within Habitats: Artificial reefs have been constructed for a variety of purposes using multiple depths, configurations, and materials. They have been constructed in depths ranging from 30 to 360 ft of water. Shallow quarry reefs (30–60 ft) off southern California have been successful at kelp recruitment, with a resultant biological zonation equivalent to that observed in natural beds (Foster and Schiel 1985; MEC 1993).

Biological Community: In most cases, reefs have been constructed to attract and support recreational or commercial fish species (Seaman and Sprague 1991). Generally, comparative studies of artificial and natural reef fish assemblages in the same area have demonstrated similarity in species composition, although species abundance and biomass may differ. Reefs have also been constructed to support the development of kelp (MEC 1993), lobsters (Collins et al. 1993), and coral (Fitzhardinge and Bailey-Brock 1989).

To date, there is no debate about the ability of artificial reefs to aggregate fish populations (Buckley 1989), but there is still considerable uncertainty in the literature regarding functional equivalency of introduced habitats (NRC 1994). Ambrose (1994) argues that data from a constructed reef in southern California (Torrey Pines Artificial Reef) support the notion that fish production is enhanced by artificial reef construction. Other authors (Alevizon and Gorham 1989; Seaman and Sprague 1991) are not as equivocal and indicate that evidence of life cycle functions associated with natural reefs have not been similarly displayed at constructed sites. At least one author (Polovina 1989) suggests that artificial reefs are nothing more than benthic fish aggregators.

By virtue of the depths in which reefs are commonly located, the foundation species are typically algae in the photic zones or colonial invertebrates (bryozoans, corals) in the sub-photic zone. In shallow-water Pacific coast reefs, the dominant algal species is frequently either *Macrocystis* spp., *Nereocystis* spp., or *Laminaria* spp. with some mixed red algae. These "artificial" kelp communities, when established, appear to support the same resources and community structure as do natural reefs (MEC 1993).

Key Ecological Processes: Important physiological and biological processes acting within artificial reefs in estuaries are not dissimilar from those within natural substrate. Proper design considerations during construction planning can maximize use of stable environmental parameters, while minimizing potential disturbances. While NOAA (1990) has advocated a more standardized approach to reef construction, it may be prudent to design a system based on specific targeted species. Light, temperature, and salinity requirements for artificial kelp beds will be different relative to reefs designed to recruit and enhance recreational rockfish catches.

Nutrient Sources and Distribution: Sources of nutrients and nutrient cycling are similar to those described for natural rock substrate.

Detrital Processing and Nutrient Regeneration: Where artificial reefs result in increased algal production, there is a net positive influx of detrital material into both the reef and the nearby soft bottom environments. The increased food source not only contributes to the reef-dwelling epifaunal populations but also to the soft-bottom-dwelling epifauna and infauna.

Habitat Heterogeneity: Artificial benthic habitats can vary significantly in design and purpose. Bohnsack et al. (1991) list the following seven elements as important design elements when planning reef building programs:

- **Material composition**—Selecting the right materials for the target species
- **Surface texture**—Rough vs. smooth surfaces
- **Shape, height, and profile**—Three-dimensional structure of the reef
- **Hole size**—Size, number, and diversity of hole sizes
- **Reef size**—Volume, bottom coverage, and surface area of the reef
- **Reef scale**—Spatial coverage of reefs within a given geographic area
- **Dispersion**—The arrangement of materials within or between reefs.

Key Natural Disturbances: Improper reef siting can lead to the placed structure sinking into soft bottom. Excessive siltation may result in the burying of the placed structure, while placement in high-energy sand environments can result in scouring of the substrate surface. Careful consideration of location and construction can minimize these concerns. The physiological (i.e., light, current, and salinity) needs of the targeted species must also be considered during the design phase. Finally, biological competition or predation must be considered during reef design. For example, kelp restoration programs have required predator control on both sea urchins and fish prior to successful plant establishment (MEC 1993).

Functional Values: Traditionally, artificial reefs have been placed to attract fish species for recreational or commercial purposes. Where there is little hard bottom habitat in an area, artificial structures provide substrate for algae and sessile invertebrates that, in turn, support fish assemblages. More recent reef designs have included long-term recruitment as mitigation for near-shore construction programs (Hueckel et al. 1989). Artificial reefs are also constructed for kelp forest or coral reef regeneration.

Causes for Deterioration: Generally, while constructed habitats physically remain where placed, they often fail to support the intended species or assemblage. Improper siting of the structures can result in conditions that are physically inappropriate for the target species or may result in it being outcompeted by a resident dominant species (Hueckel et al. 1989).

Assessment of Habitat Health: Much of the literature on "habitat health" for artificial reefs focuses on methods of assessing the relative success of the habitat. Where specific objectives were enumerated prior to construction, assessing habitat health is a function of comparing results to expectations. For reefs placed simply to enhance recreational fisheries, the assessment involves questioning fishermen on the results of their catch efforts in reef vs. non-reef areas. Where long-term habitat is the objective, traditional ecological assessments and monitoring tools, which involve measurements of community structure and function and changes in structure and function with time, are required (Bortone and Kimmel 1991).

In-Bay Terraces—In-bay terraces are a relatively new type of in-water restoration project that have shown considerable promise in creating both hard and soft bottom estuarine habitats. Terracing creates a flat, or gently sloping, habitat that is shallower than the deeper area it replaces. Subtidal rock retaining walls are constructed along the terrace perimeter, the site is back-filled with dredged sediments, and then clean material suitable for the target restoration species (e.g., sand for eelgrass transplant programs) is placed over the dredged sediments. The sediment-capped areas have been used to establish eelgrass (*Zostera marina*) meadows and shallow water habitat for foraging by the California least tern (*Sterna antillarum browni*) (MEC 1993).

In-bay terraces also have been used in conjunction with disposal of contaminated soils or sediments. These confined aquatic disposal (CAD) sites are designed and constructed both as remediation and restoration structures. The contaminated sediments are placed at the base levels of the terrace behind diked walls and then capped with clean material, both confining the contaminants and providing suitable clean material for recolonization by benthic infauna or submerged aquatic vegetation.

Geographic Distribution: To date, the terracing programs have primarily been constructed in southern California, specifically in the harbors of San Diego, Long Beach, and Los Angeles. Two programs from the region demonstrate the restoration potential. The most successful example of terracing for purely restoration purposes is the Le Meridien Eelgrass project in San Diego Harbor. This program involved constructing a 180-ft–long quarry rock retaining wall and backfilling with clean sand to an elevation of −6.5 to −7.5 ft mean low water (MLW) to create a total surface area of 7,000 ft^2 for transplanting eelgrass

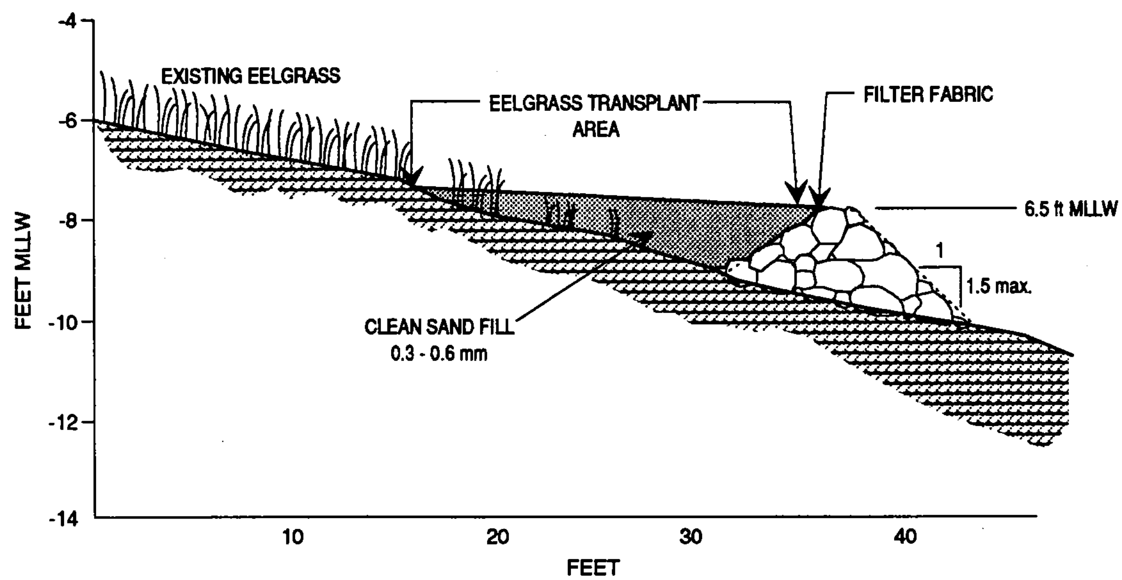
(Merkel and Hoffman 1990). A schematic diagram of the site's construction is shown in Figure 5B-9. In the 5 years since construction and transplantation, the eelgrass meadow has been successful at duplicating functions of natural beds (Merkel 1995, pers. comm.). An added, unexpected benefit was the value of the quarry-rock dike as an artificial reef. Constructed to a total height of more than 20 ft, the reef has attracted fish (e.g., sand bass, opaleye, midshipmen, surf-perch), invertebrates (e.g., spiny lobster, octopus), and seaweeds (e.g., *Sargassum*).

An example of a combined CAD and restoration program is the Pier 400 permanent shallow water habitat (PSWH) at the Port of Los Angeles (Pier 400 Design Consultants 1995). The PSWH is located on the harbor side of the San Pedro Breakwater (Figure 5B-10), and when complete, it will be a 190-acre shallow water area. Within the PSWH is a 94 acre CAD site that is filled to a depth of −30 in. MLW with contaminated sediments and then covered with an additional 15 ft of clean dredged material. The final 2 ft of the site is finished with clean sand. The PSWH dike retaining wall is constructed with quarry rock to a height of approximately 25 ft. When completed, the rock wall should provide habitat for species assemblages similar to the Le Meridien Eelgrass project. The submerged terrace sand surface was designed to provide optimal foraging habitat for *S. antillarum browni*.

Biological Community: As with other forms of artificial habitats, terraces are designed for specific species or assemblages, but they also yield additional benefits for other non-target species. For example, the Le Meridien Eelgrass terrace supports life functions for a number of fish species, including top smelt, shiner and black surfperch, gobies, and pipefish. Numerous macroinvertebrates also use the eelgrass bed as nursery and/or foraging habitat. The habitat also provides bird forage, including *S. antillarum browni*.

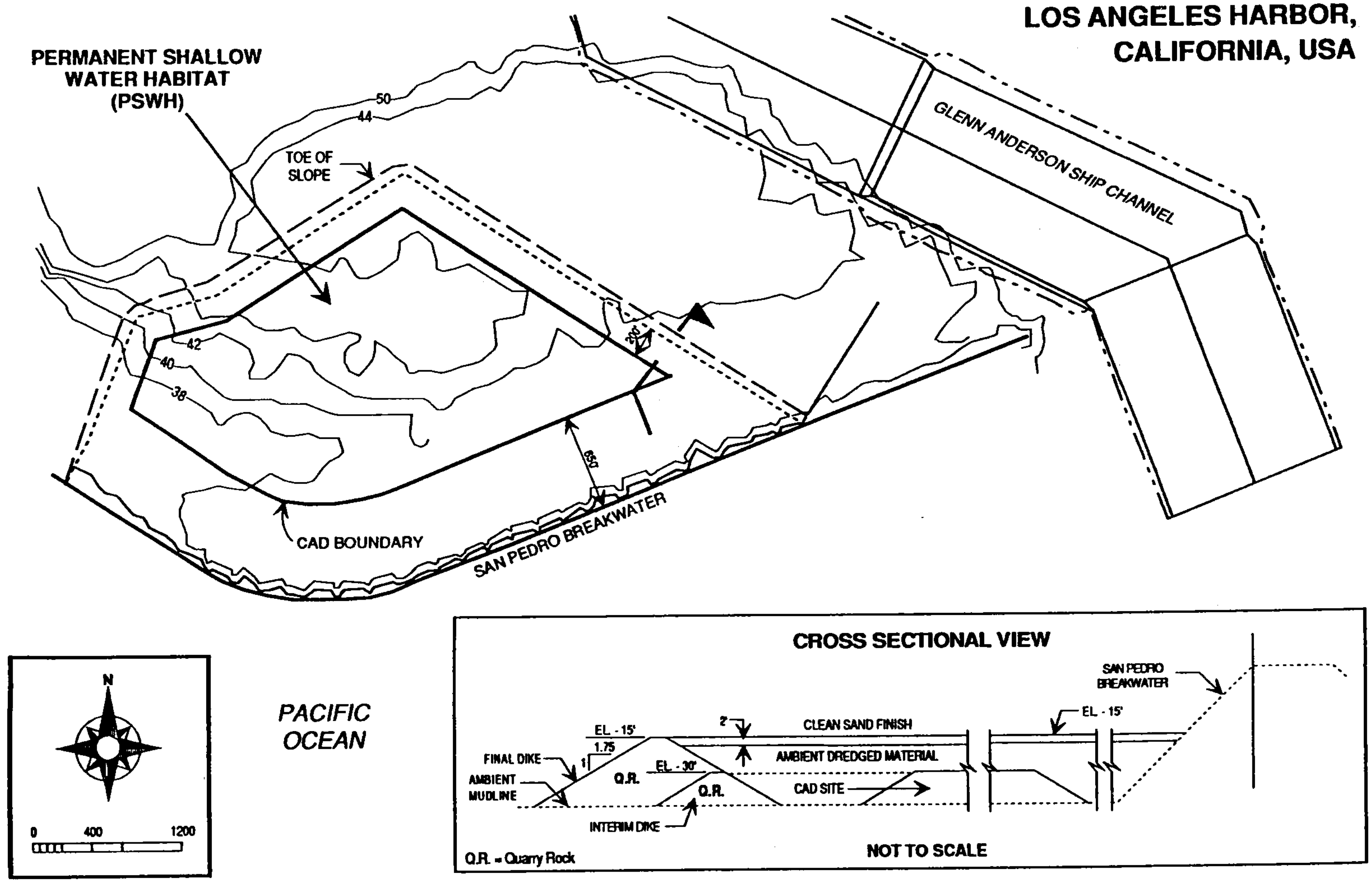
An earlier terracing project for the Port of Los Angeles, California, the Pier 300 Shallow Water Mitigation project, created a 190-acre clean, sandy substrate that appears to provide nursery areas for queen fish, northern anchovy, California grunion, and California halibut (MEC 1993). The site also had the highest bird usage, compared to other harbor areas, and was used by *S. antillarum browni* as a foraging location.

Key Ecological Processes: The planning and design considerations discussed under artificial reefs are applicable to terrace projects, along with the species- or assemblage-specific habitat requirements. When coupled with CAD, the design should consider appropriate safeguard measures for sediment containment to ensure effective isolation of the chemicals of concern.



Source: Reproduced with permission from Merckel and Hoffman (1990)

FIGURE 5B-9. Cross sectional view of Le Meridien Eelgrass Restoration Area.



Source: Pier 400 Design Consultants (1995).

FIGURE 5B-10. Permanent shallow water habitat and confined aquatic disposal (CAD) site for the Port of Los Angeles Pier 400 project.

Modified Substrates

Substrate modification is the placement of natural materials that may alter the relief of the basin or channel bottom and significantly alter texture. These types of projects usually consist of placing shell, gravel, or rock mixtures in intertidal or subtidal areas to provide habitat for benthic and epibenthic organisms.

The oldest and most frequent substrate modification programs in the United States have involved the creation of areas to encourage the settlement and recruitment of oyster spat in the Atlantic and Gulf coast estuaries. These "oyster reefs" have been built by the U.S. Army Corps of Engineers (Corps) in both the Galveston and Chesapeake bays by using dredged material to raise the bottom elevation and covering the new areas with suitable cultch or gravel.

More recently, efforts have been undertaken to provide similar seafloor modifications to restore lost crab or fish habitat. The Corps Seattle District undertook a program of placing 9,000 yd^3 of shell hash, 3-in. thick, over 20 acres of mudflats in Grays Harbor, Washington, to provide habitat for Dungeness crabs (*Cancer magister*) (Corps 1989). The Port of Tacoma, Washington, undertook a program to terrace the slope of a shoreline from a 2:1 to a 5:1 slope using clean, dredged material and then placed gravel mix over the fill to encourage the development of epibenthic organisms that were the preferred prey items for juvenile salmonids (Jones and Stokes 1991).

Geographic Distribution—Oyster reef construction has been most actively practiced in the Gulf coast estuaries and in Chesapeake Bay. Cobble/gravel shoreline modifications for salmonid enhancement have apparently been limited to the Pacific Northwest.

Biological Community—Like artificial reefs, substrate modifications are generally designed to enhance or restore a specific species or assemblage of species. While oyster reefs in the Gulf and Atlantic coast estuaries are constructed to restore or enhance *Crassostrea virginica*, this hard bottom substrate also supports other bivalves (e.g., mussels, cockles) and a variety of epibenthic organisms (e.g., polychaetes, amphipods). Larger, commercially and ecologically important species in turn prey on the bivalves or epibenthos. These species include blue crabs (*Callinectes sapidus*), fish such as the naked goby (*Gobiosoma bosci*), and terrestrial animals that prey on exposed reefs during ebb tides (e.g., raccoons, wading birds) (Bahr and Lanier 1981).

Key Ecological Processes—Artificial habitats of placed gravel or cobble are subject to the same physical and biological pressures described for natural habitats. There are some unique design considerations in creating these modified habitats over an essentially soft sea floor. Estuarine soft bottoms are generally depositional or erosional in character. The most important cause of failure in substrate modification programs is erosion and/or "silting-in" of placed materials. For example, in the Grays Harbor, Washington, crab habitat project, oyster shells placed for habitat modification have consistently become covered with sediment (Armstrong et al. 1991). The gravel placed in the Port of Tacoma, Washington, was eroded by winter storm action at one end of the site, while deposition and burying of existing gravel occurred in other portions. To reduce erosion, larger-sized fill may be placed at, or on top of, the erosional zone in a process called "armoring." Sedimentation or sinking is more difficult to control once a structure has been placed; however proper siting studies can limit those problems.

In some instances, creating a structure on an otherwise nearly level bottom will offer two advantages. First, the top of the structure may receive more light than the surrounding seafloor, resulting in more suitable conditions for algae production. Second, the structure may create turbulent flow conditions that will prevent the accumulation of sediment on the structure's surface. The specific design features that will determine whether the structure's surface is likely to accumulate sediment are very important and include the following:

- The dimensions of the structure relative to the width and depth of the water body within which the structure would be placed
- The magnitude of the current moving past the structure
- The surface roughness of the structure
- The slope of the sides of the structure.

KEY ENVIRONMENTAL PARAMETERS

The key environmental factors for subtidal soft bottom restoration projects depend on the nature of the proposed restoration. Table 5B-4 summarizes key environmental parameters that must be considered in implementing restoration projects in subtidal estuarine habitats. The concept of restoration usually assumes that the habitat has changed from a baseline condition, and the goal of the project is to restore the habitat of interest back to the original baseline conditions (NRC 1994). If pre-existing baseline conditions have not been documented, an adjacent reference habitat may be used as a model for identifying key parameters. The 12 habitat parameters listed above in *Assessment of Habitat Health*, or a subset of these, can be used to compare the potential restoration site with a baseline or an adjacent model. In addition to the 12 habitat parameters, one needs to consider

TABLE 5B-4. KEY ENVIRONMENTAL PARAMETERS IN SUBTIDAL ESTUARIES

Parameter	Comment
Physical	
Sediment deposition	Sediment is trapped on the continental margins and land is accreted; outlets for drainage/filtration from continental margin to the open shelf are diminished; flood control (decrease floodwater depth) by dispersing water over a large area; increase area covered by land-water interface (interfaces are biologically important); coarse-grained sediment intercepts and attenuates wave and current turbulence.
Chemical	
Sediment chemistry	Oxidizing and reducing gradients are established in sediments that influence sulfate reduction, redox position, metal solubilities, inventories of ammonia, sulfide and redox sensitive toxics; binding of contaminants to sediment particles; chemical precipitation (metal sulfides and oxides, carbonates, contaminants); solid-dissolved interactions.
Nutrient recycling	Organic materials are remineralized.
Biological	
Substratum enhancement	Substratum in the subtidal estuarine habitat provides sites for attachment (benthic, other); provides food for the detrital food web; provides shelter/refuge for benthic organisms.
Particle size distribution	Coarse-grained sediments are better for certain fish because fine-grained sediments can clog gills. Fine-grained sediments are better for infaunal detritivores; firmer, coarser sediments are better for filter feeders requiring firm substrates. Sediments can affect water quality and ecosystem health; fine-grained and organic-rich sediments are potential reservoirs for contaminants; bioturbation, suspension feeding, as it affects sediment character (porosity, stability, water content, oxygenation, organic content, rate of bio-deposition); trapping of fine sediments by submerged plants and diatom mucus films; biological precipitation (coral, algal reefs and molluscan shell deposits)
Grazing/foraging/predation	Predator/prey or grazing/plant imbalance resulting in loss of the base of the food chain; loss of diversity/stability; loss of market basket species or aesthetic value (reefs).
Natural Disturbance	
Weather and climate	Storms; floods; ice scour (cool-temperate and arctic regions); geological (i.e., tectonic [earthquakes, subsidence, gravity-slumping, volcanic ash/flows], sedimentological (filling due to natural sedimentation rates]); sea level change (component due to natural warming and deglaciation; isostatic rebound).

TABLE 5B-4. (cont.)

Parameter	Comment
Human Disturbances and Alterations	
Human encroachment	Prevention structures created to control flood and storm damage can disturb or alter subtidal estuarine habitats. Other encroachments include: maintaining navigable channels and harbors; dredged material disposal; alterations for agricultural purposes (e.g., drainage, irrigation); use of natural resources (fishing, shellfishing, mining, logging, oil and gas development); recreation (beach maintenance, navigation); health, sanitation, and circulation (siting of inflow, outflow pipes and channels, drainage of wetlands, mosquito control practices); construction of wharfs, docks, breakwaters, seawalls, groins, revetments, bridges, causeways, roads, underwater utility lines, levees, riprap, moorings also effect the subtidal habitat. Types of disturbances and alterations: habitat loss or alteration; removal of bottom material (dredging, borrow pits, channelization); addition of material to bottom (dredged material disposal, construction debris disposal, other kinds of disposal, chemical or radioactive dumps, artificial reefs); altering circulation (channel alteration, straightening, drainage canals, creating orthogonal patterns, altering bottom friction, dams, floodgates, tidegates); contaminant and nutrient inputs (oil and chemical spills, effluent and point-source discharge, non-point source runoff); other physical disturbances (fisheries activities such as intensive cage mariculture, raking, or bottom trawling, mining or drilling activities, wrecks); thermal changes (thermal effluents from power plants); global warming (affects coral reefs, climate, sea level rise, productivity).

other physical-chemical parameters, including bottom sediment stability, sediment texture, geotechnical properties such as sediment water content and cohesion, contaminant inventories, salinity regime, bathymetry, water turbidity, and light penetration. The list of key environmental parameters is potentially large. It is, therefore, important to identify a subset of these parameters that are relevant to the specific restoration project under consideration. The key environmental parameters to consider for subtidal soft bottom restoration are as follows:

- **Disturbance Regime**—If restoration involves the creation of bottom structures or bottom sediment remediation, the frequency and intensity of both natural (storm) and anthropogenic (e.g., trawling) disturbance is important. Sediment remediation may only be effective where major storms do not significantly resuspend the bottom or where bottom trawling is infrequent.

- **Sediment Quality**—Because sediment quality is a major factor controlling benthic succession, the rate and extent of habitat restoration may be limited by this important variable. Sediment quality includes not only the types and concentrations of contaminants and their bioavailability but also physical factors such as granulometry and geotechnical properties (e.g., water content, physical stability).

- **Water Quality**—The kinds and numbers of species and trophic types that may naturally colonize an area of remediation will be sensitive to the salinity regime (e.g., Table 5B-1). Ambient turbidity at, or near, the sediment-water interface is an important factor for suspension feeders because suspended loads of very fine particles may clog filtering mechanisms and exclude this trophic group. This key factor is usually linked to bottom stability (Rhoads and Young 1970). All commercially important bivalves are suspension feeders, so this factor is important to consider for enhancing bivalve productivity. Suspended seston also is a key factor for light penetration and is a key factor for determining the distribution of SAV. Dissolved oxygen concentrations are critical for maintaining secondary and higher productivity, especially for high metabolic rate species such as lobsters, crabs, and fish (Tyson and Pearson 1991).

RESTORATION PROJECTS

Potential restoration projects in subtidal estuarine habitats are summarized in Table 5B-5. The first column lists problems that may cause habitat loss, damage,

TABLE 5B-5. RESTORATION PROJECTS IN SUBTIDAL ESTUARIES

Problem	Possible Solutions
Disturbed habitat	Disturbance management: Capping is the placement of clean sediments or rocks over material that must be kept in place, sealed, and sequestered to avoid dispersal (e.g., contaminated dredged material deposits, chemical or munitions dumps). Considerations in evaluating feasibility of capping include: water depth, bottom topography, shipping, currents, dredged material and capping material characteristics, and site capacity (Palermo et al. 1989). Materials used for the uppermost layer of capping should be sand or stones to minimize sediment transport offsite. The layer below ideally would be finer sand or coarse silt to minimize pore space and movement of contaminated fluids out of the contaminated disposal mound beneath, although if the underlying material is weak or water-rich, there may be temporary instability. Organic-rich sediments, even if otherwise uncontaminated, should be avoided as a capping material in estuarine environments because excess nutrients can leach out of the capping material and potentially contribute to eutrophication. Capping has been accepted by the London Dumping Convention as a physical method of rapidly rendering harmless contaminated disposed material in the ocean. Engineering aspects of cap design and placement are addressed by the U.S. Army Corps of Engineers' Dredging Research Program (Palermo et al. 1989). Isolation measures include capping (discussed above) and subaqueous confinement measures such as use of borrow pits or construction of subaqueous dikes to confine materials. The latter method has been used in Europe and to a limited extent by the U.S. Army Corps of Engineers New York District to confine disposal of material during a disposal operation. The effectiveness of subaqueous confinement is increased if materials are placed precisely and submerged points of discharge or submerged diffusers are used. The area should have enough volume for storage of settled and consolidated material and be a natural sedimentation site, rather than erosional. The supernatant water is discharged over a weir or other energy-absorbing device, designed to minimize "splashing" or resuspension of already settled material (Palermo et al. 1989). Submerged diffusers have been used in the Netherlands and in the United States (Hayes et al. 1988). For many millennia, man has physically manipulated the natural environment to enhance productivity. Primitive cultures have used slash and burn or periodic flooding, while more advanced cultures use selective timber cutting or plowing. These same principles can be applied regarding management of subtidal benthic habitats. Providing dredged materials are free of contaminants, disposal frequency can be managed to maintain the area of disposal impact in log phase colonization. Pioneering stages tend to have high rates of population growth and high population density and are readily available as prey for higher trophic levels. Rhoads et al. (1978a) show that pioneering stages of dump site recolonization, consisting mostly of tubicolous polychaetes, are at least 6 times more productive than ambient bottom production. New space created by disposed dredged materials offer non-competitive space for colonizers, organic-rich detrital food, and spatial refuge. The Benthic Resource Assessment Technique, when applied to the Massachusetts Bay Disposal Site, found that the gut fullness of the American Plaice was 5 times higher on disposal mounds than on the ambient bottom (SAIC 1987).

TABLE 5B-5. (cont.)

Problem	Possible Solutions
Contamination of habitat and destabilization of soft-sediment features	Restoration and rehabilitation: Removal and decontamination, replacement with uncontaminated sediments, capping (e.g., New Bedford Harbor Superfund polychlorinated biphenyl remediation); preventing or minimizing further contamination. The feasibility of stabilizing soft-sediment features in the subtidal regime of estuaries depends on whether the environment is primarily depositional or erosional and whether the desired restoration is based on containment or dispersal of material. If erosive forces are continually present, then restoration activities such as addition of material to an eroding subtidal sand bank may be doomed to fail; on the other hand, if erosion is occasional or intermittent, then such restoration may be worthwhile.
Destruction of reef habitat	Creation and replication: Artificial reef construction (materials have included coal combustion wastes [Carleton et al. 1982]). Restoration or creation of suitable habitat for fish and shellfish must consider species and life-stage of interest; its suitability to the habitat proposed; and its needs in terms of tidal amplitude, circulation, salinity facies, oxygenation, organic inputs, preferred substrate, sediment transport, potential predators or parasites already present, presence of associated species (vegetation, animals) required for long-term success, tolerance to local conditions, and prior history of that species in the area. Furthermore, the amount of time spent by the species in the habitat proposed must be considered. Fish that breed and/or spawn and whose juvenile stages spend their early life in estuaries generally prefer well-aerated habitats with soft but not muddy sediments (that can clog gills), submerged and/or emergent aquatic vegetation for food and cover, and a range of salinities. Other fish, such as salmon, may prefer hard bottom, rocky, or stony substrates that offer aeration and crevices for cover. Shellfish habitats include subtidal to intertidal sand and mudflats and tidal channels (clams and oysters), hard bottom (mussels), muddy bottoms (lobster, crawfish, shrimp). Such soft bottom habitats may be ephemeral on a longer time scale or easily disturbed by natural or human disturbances.

or deterioration in each type of habitat. For each problem, one or more possible solutions is given in the second column.

The goal of most subtidal soft bottom restoration projects is to restore/enhance secondary productivity or to prevent contamination of an already productive system. Site selection criteria for containment management dictates that dredged materials be placed within a low kinetic energy environment, which is commonly a subtidal soft bottom area. The effect of dredged material disposal on secondary benthic production was first pointed out by Rhoads et al. (1978), who showed that pioneering macrofauna enhanced secondary productivity on disposal mounds 6 times relative to that on the ambient seafloor. The enhanced productivity and structural complexity of disposal mounds were proposed as factors to explain the common observation that disposal mounds attract both lobsters and demersal fish. To maintain high secondary productivity, disposal needs to be repeated periodically to keep the system in a pioneering stage so that succession does not progress into less productive mature assemblages (Table 5B-2). The idea of managing a disposal site as a pulse-stability system was borrowed from early ideas expressed in Odum (1969) regarding compromise ecosystems.

Once disposal sites were recognized as trophic and structural "magnets" for predators, sediment quality became a topic of concern. Disposed sediments that are of management concern are candidates for special management options, including capping with clean sediment to prevent food chain contamination. Much of the pioneering work on restoration of habitat quality and productivity by means of capping of subtidal disposal mounds was done in New York and New England waters under the direction of the Corps New York District and New England Division. A summary of this experience has shown that capping has been effective in isolating buried particulates and pore waters from the sediment-water interface, even with the passage of several hurricanes (Murrey et al. 1994; Stivers and Sullivan 1994; May et al. 1994). The Seattle District of the Corps has also pioneered capping in the Puget Sound, Washington, area. A high profile project involved capping 217,000 m^2 of creosote-contaminated sediment in Eagle Harbor with a 0.9-m-thick layer of sand. The sand was gently spread over the bottom to limit bottom disturbance, which had the potential for remobilizing creosote. The physical monitoring program has shown this tabular cap to be uniformly placed and effective in isolating creosote from the overlying water column. Biological investigations are in progress to assess recolonization and bioaccumulation (Nelson et al. 1994).

The way in which disposed materials are placed and oriented on the bottom can influence the value of the bottom for fisheries. For example, by controlling the size, height, geometry, and orientation of the disposal berms in the Gulf of Mexico, disposal sites have been shown to attract fish, particularly in the lower kinetic energy lee-side current shadow. The berms have been shown to have a higher percentage of shrimp, an important food resource for red snapper, than the ambient bottom (Clarke and Kasul 1994).

Estuary models are generally well developed and involve three model types: 1) food web models, discussed above in the context of biological communities; 2) structural models; and 3) dynamic models. The approach to structural and dynamic modeling described below can clearly be applied to other types of aquatic ecosystems, as described more fully in SAIC (1996b). The structural models shown in Figures 5B-3 through 5B-5 show relationships but not dynamics. Dynamic models require a significant increase in effort over food web models to quantify fluxes. Two classes of dynamic box models exist: energy flow models and dynamic ecosystem models. Classic work on energy flow models has been carried out in Georgia salt marshes, and studies are summarized in Montague et al. (1981). The unit of energy flow is usually based on carbon, but nitrogen can also be modeled if the emphasis is on protein yield. The boxes in the model represent temporary storage sites, and interconnecting pathways represent fluxes (Figure 5B-11). Experimental measurements are useful to quantify linkages between biotic components and fluxes, or inferences are made about energy flows by tracing stable isotope ratios through the system of interest. Energy flow models are useful in habitat restoration models if the main purpose of the project is to restore productivity and reestablish a commercially important food chain.

Dynamic ecosystem models such as the coupled differential equation (CDE) model (also known as the Ems-Dollard ecosystem model) require even more input variables than an energy flow model because physical transport processes such as advection, diffusion, sedimentation, resuspension, and residual sediment transport are included (Baretta and Ruardij 1988). Abiotic variables and processes such as sunlight, temperature, salinity, nutrients, silt content of the water column, and pelagic and benthic oxygen are quantified. Biological processes include all key physiological functions driven by photoperiod, temperature, and reproductive periodicity. Because of the complexity of dynamic ecosystem models, teams of scientists are required to quantify the inputs, run the model, and verify or validate the model with field data. The CDE model has been run for Narragansett Bay and the Ems-Dollard Estuary in the Netherlands. A critique of the CDE model, as applied to these two estuaries, has been made (Rothschild Ault Group 1992). Generally, the CDE model performs well in predicting average annual magnitudes of all variables, particularly those regulated by physical forcing functions and physical/abiotic processes. The poorest correspondence between simulated and observed values was for those variables that were mostly dependent on biological variables. The CDE model is expensive to set up and is available for only one estuary in the United States (Narragansett Bay). Therefore, this approach has only limited application to habitat restoration at this time. However, as other estuaries are similarly modeled, this approach should be invaluable for running simulations on different restoration scenarios to evaluate impacts on the overall system.

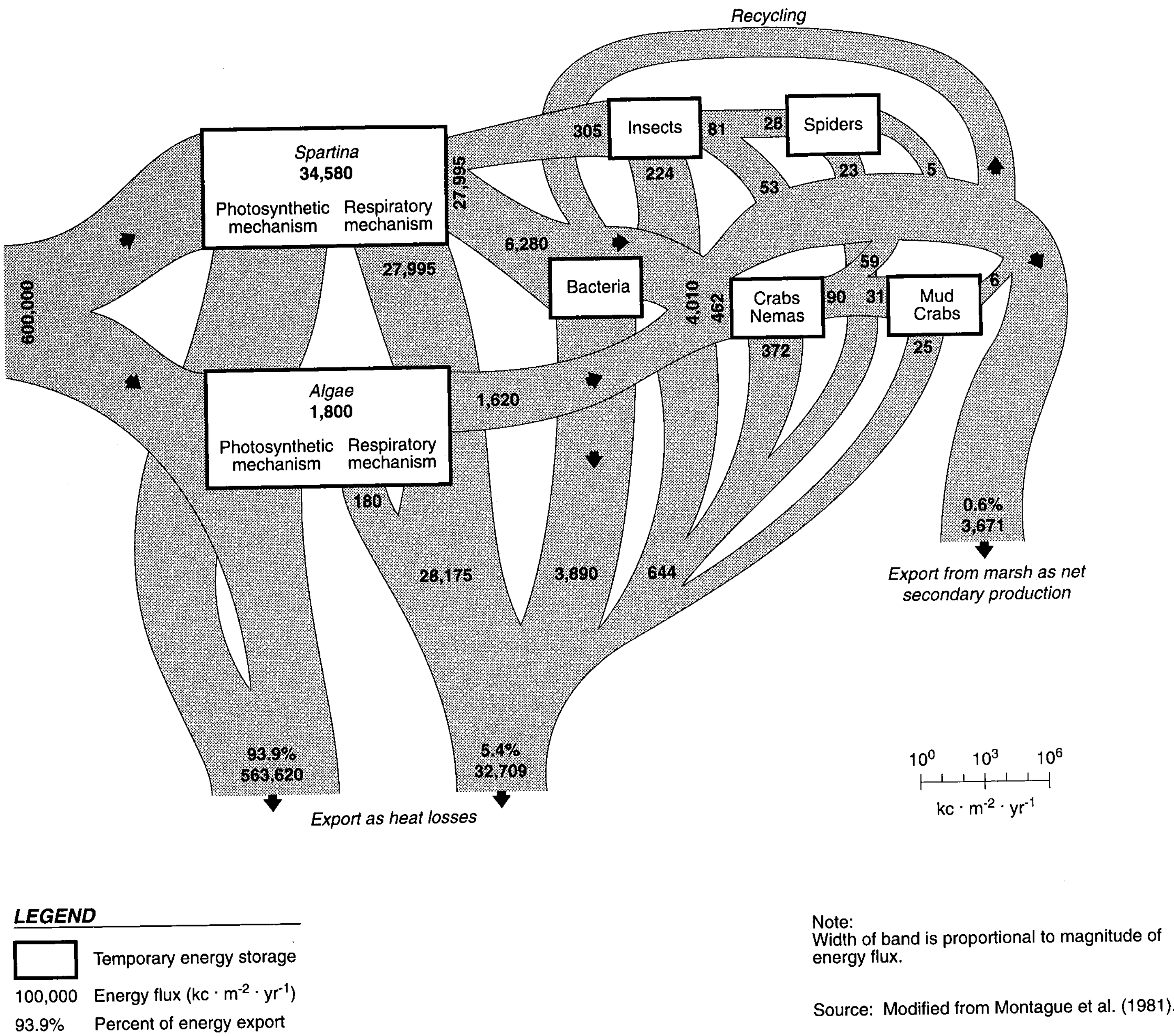


FIGURE 5B-11. Example of an energy flow diagram based on carbon flux for a Georgia salt marsh.

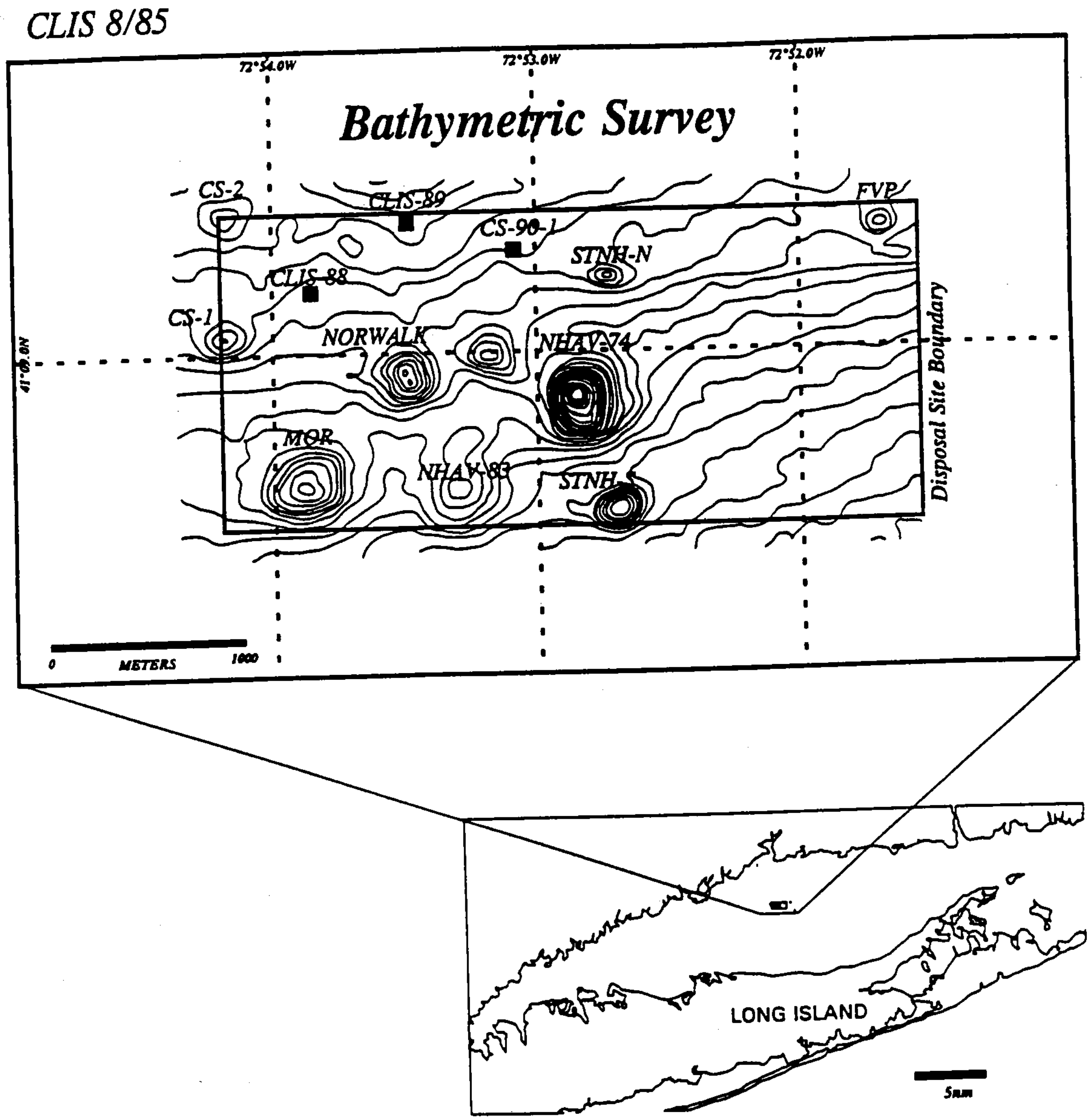
Three case studies are presented below to illustrate restoration projects in subtidal estuaries. Capping has become a popular management strategy for improving benthic habitat quality at dredged material containment mounds where elevated concentrations of metals, pesticides, or hydrocarbons can occur. The first case study, capping of dredged materials at Long Island Sound, is an example of this strategy. Capping of *in situ* contaminated sediments is also a method of reclaiming benthic habitat quality where dredging and removing sediments in not a good option because of the risk of far-field dispersion of contaminants. The Wyckoff/Eagle Harbor case study is an example of this application. Finally, the third case study describes an effort to dispose of dredged material and construct a substrate that would be resistant to sedimentation. This project took place within Slaughter Creek, a part of the Chesapeake Bay estuary characterized by natural oyster bars.

Capping of Dredged Material Containment Mounds in Long Island Sound

Since 1979, the Corps New England Division has pioneered the use of capping to manage dredged material that contains contaminants derived from port and channel dredging in Connecticut and New York waters. Dredged material is barged to disposal sites within the Central Long Island Sound (New York) Disposal Area and deposited at taut-wire reference buoys to form discrete mounds (Figure 5B-12). Capping is then used to isolate contaminants from the ecosystem, thus ensuring physical and chemical stability of capped contaminants over long periods of time. Four of the mounds within the disposal area have required capping and long-term monitoring (SAIC 1995).

The mounds within the disposal area are located on a topographically flat (i.e., level) silt-clay bottom ranging in depth from 20 to 25 m. No submerged aquatic vegetation exists at these depths. This habitat represents a relatively low kinetic energy environment where fluid shear stresses from surface waves rarely reach the bottom. Because most dredged materials from harbors also contain a significant fraction of silt-clay, the Corps has historically tended to dispose mud over mud to ensure long-term containment. The region has commercially important lobster and flounder populations associated with the ambient silt-clay bottom that support a local fishing industry. These commercial species prey on infaunal worms, bivalves, and crustaceans. Ambient sediments surrounding disposal sites tend to have low background concentrations of metals and hydrocarbons and are highly bioturbated and aerated. However, during late summer and fall, hypoxic water can enter this part of the sound from eutrophic waters to the west.

The major objective of capping dredged materials is to isolate the ecosystem from contaminants. To achieve isolation, a cap must be created that is sufficiently thick to prevent penetration of it by deep burrowers and that prevents or limits upward diffusion of soluble contaminants in pore water (Murray et al. 1994).



Source: SAIC (1995)

Note: Each mound (N=13) represents a separate project for management purposes. Not all of these mounds are capped.

FIGURE 5B-12. Long Island Sound and the central Long Island Sound disposal site.

The cap must also resist storm erosion over long periods of time. These management objectives are necessary because disposal site mounds tend to attract commercially important species because they provide habitat complexity (the berm or "reef" effect) and because they are prey-rich due to the fact that they tend to have higher secondary production than the ambient bottom (Rhoads et al. 1978) and attract fish to the lee or current shadow side through berm-flow interactions (Clarke and Kasul 1994).

All of the capping projects in Long Island Sound were planned and implemented by the Corps New England Division as part of the Disposal Area Monitoring System (DAMOS) Program.

Restoration Approach

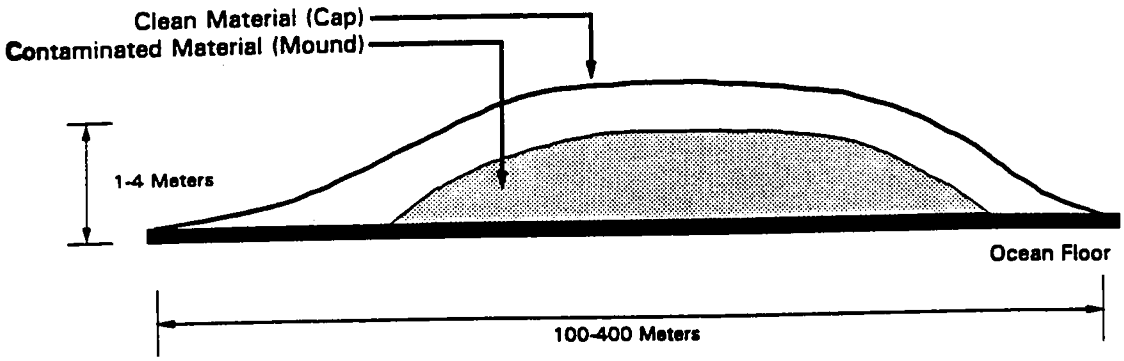
The decision to cap or not to cap a site is based on extensive testing of sediments prior to their removal. A tiered testing protocol is used to arrive at this decision. If bulk sediment chemical analysis raises a red flag, bioassay and bioaccumulation tests may be run for further evaluation (Germano et al. 1994). Early project planning includes running a DAMOS capping simulation model, which predicts the mound footprint and height based on dredged volume and other salient parameters (Wiley 1994). This same model is used to estimate the quantity of clean sediment (usually sand) required to cover contaminated sediment with a critical thickness of capping material (Figure 5B-13). Finally, the dredging schedule observes certain biologically sensitive periods to avoid conflict with spawning, migration, and recruitment of local species.

Once the project is completed, the project footprint, height, cap thickness, and lateral continuity is documented with detailed acoustic and optical surveying techniques. Long-term monitoring is conducted each year (especially after passage of major storms) to ensure cap stability.

Evaluation of Restoration Efforts

The success of capping projects is evaluated by three criteria: adaptive management, ecosystem-level planning, and fail-safe success. These criteria are described below.

Adaptive Management—Dredged material capping, as currently practiced, has been preceded by a long history of experimental projects, with periodic review of results by the Corps New England Division's Technical Advisory Committee (SAIC 1995). Results are also presented at public meetings that



Source: SAIC (1995)

Note: Typical Long Island Sound water depths range from 20 - 25 m.

FIGURE 5B-13. Schematic section of a capped mound.

involve interested citizens and government agencies (e.g., EPA, National Oceanic and Atmospheric Administration, and state regulatory entities) as well as private organizations (e.g., Sierra Club, Oceanic Society). Through a process of iteration and improvement in methodologies, capping has generally been accepted as a valid management technique for contaminated sediment management.

Ecosystem-Level Planning—Early work on disposal site management was focused entirely within the perimeter of the disposal area, as designated on hydrographic charts. As long as projects conformed to specifications and disposal effects were contained within the disposal area, larger-scale ecosystem issues were not a major part of management's concern. However, when low dissolved oxygen conditions in Long Island Sound became a problem over large areas of the sound in the 1980s, the Corps had to expand their perspective. Low oxygen conditions had a dramatic effect on disposal site recolonization rates and negatively impacted the reference stations used to compare ecological conditions of disposal sites relative to the ambient seafloor.

Safe-Fail Success—Long-term monitoring is an integral part of the DAMOS program (Germano et al. 1994) and involves annual physical, biological, and chemical assessments of capping efficiency. If annual monitoring indicates that the cap has physically failed, succession has been arrested (Rhoads et al. 1978), or tissues of colonizing species have been contaminated, management options, including recapping, are evaluated. Since 1979, only one of the Central Long Island Sound projects has required special management; in that case, annual monitoring showed anomalous rates of benthic colonization and a contaminated cap chemistry (Murray 1996). Long Island Sound capped sites have survived the passage of several hurricanes.

For more information on this project, contact:

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(617) 647-8291

In Situ Capping in Wyckoff/Eagle Harbor

Eagle Harbor is a small embayment on the east side of Bainbridge Island in central Puget Sound, Washington (Figure 5B-14). The harbor is rimmed with

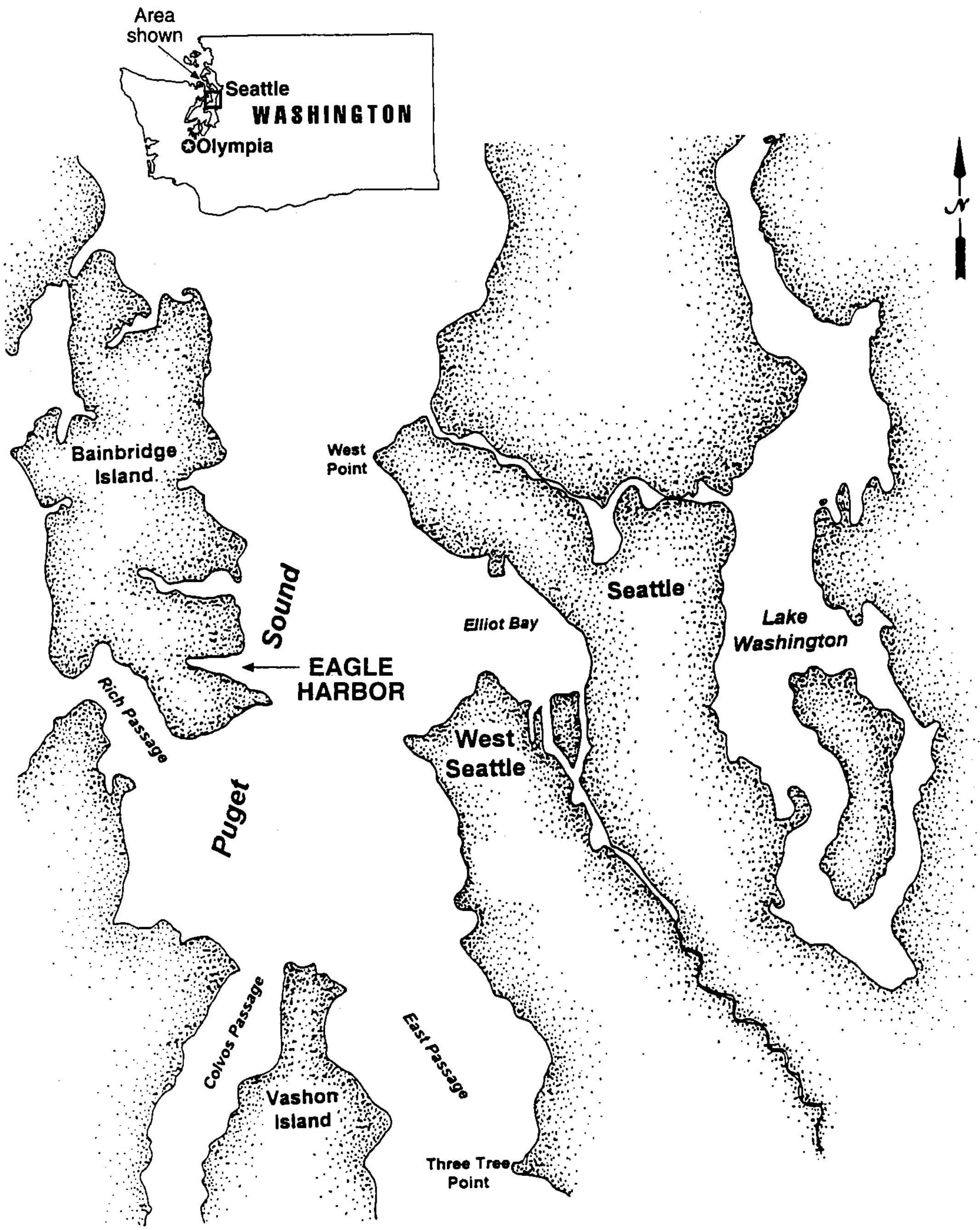

FIGURE 5B-14. Eagle Harbor location map.

residences, marinas, small boat yards, light commercial industry, and a Washington State Ferry terminal. Historically, two major sources of contaminants were located on the harbor: a wood treating facility on the south shore and a shipyard located on the north shore. Baseline monitoring documented contaminants related to these facilities in both sediments and shellfish in 1987. Based on that baseline study, a part of the harbor was placed on EPA's National Priorities List.

High concentrations of polycyclic aromatic hydrocarbons (PAHs) associated with wood treatment (creosote impregnation) and high concentrations of copper, mercury, and zinc from antifouling paints from the shipyard contaminated both sediments and shellfish. Capping of sediments was evaluated as a restoration option at a site referred to as the East Harbor Operational Unit (Figure 5B-15). Part of this area is populated by eel grass, with sandy silty muds lying between 25 and 55 ft below MLW. The site was divided into two areas to evaluate different modes of cap emplacement.

The major objective of capping was to isolate the contaminant "hot spot," which largely occupied Area 1. Goals of this *in situ* capping were to prevent contaminants from diffusing or being advected into the overlying water column and to prevent burrowing organisms from penetrating the cap. The mean burrowing depth in Eagle Harbor was determined to be approximately 10 cm. The cap was also intended to promote recolonization of the substratum by infaunal organisms so that secondary production could be restored. While these objectives are similar to those described for capping of dredged material in Long Island Sound, the methods of cap emplacement, as described below, are very different.

Restoration Approach

Sand was chosen as the capping material because well-washed sand is clean and does not disperse far from the site of dumping. A total of 275,000 yd^3 of clean, medium sand (0.25 mm) was dredged from the Snohomish River channel and disposed over 54 acres of contaminated bottom. The design thickness of the cap was 3 ft to ensure isolation of buried contaminants. In Area 1, water depths ranged from 20 to 55 ft below MLW and 125,000 yd^3 were dumped from a split-hull barge. Successive passes of the barge gradually built the cap up in increments of about 0.3 ft. The incremental approach was used to minimize disturbance of the bottom, which had the potential for massive resuspension and far-field dispersion of contaminated sediment.

In Area 2, approximately 150,000 yd^3 of sand was deposited via hydraulic pumps or washing from the barge with a water jet. Area 2 had fine-grained surface sediments and so hydraulic emplacement was used to minimize bottom resuspension from turbulence related to convective descent of masses of sand.

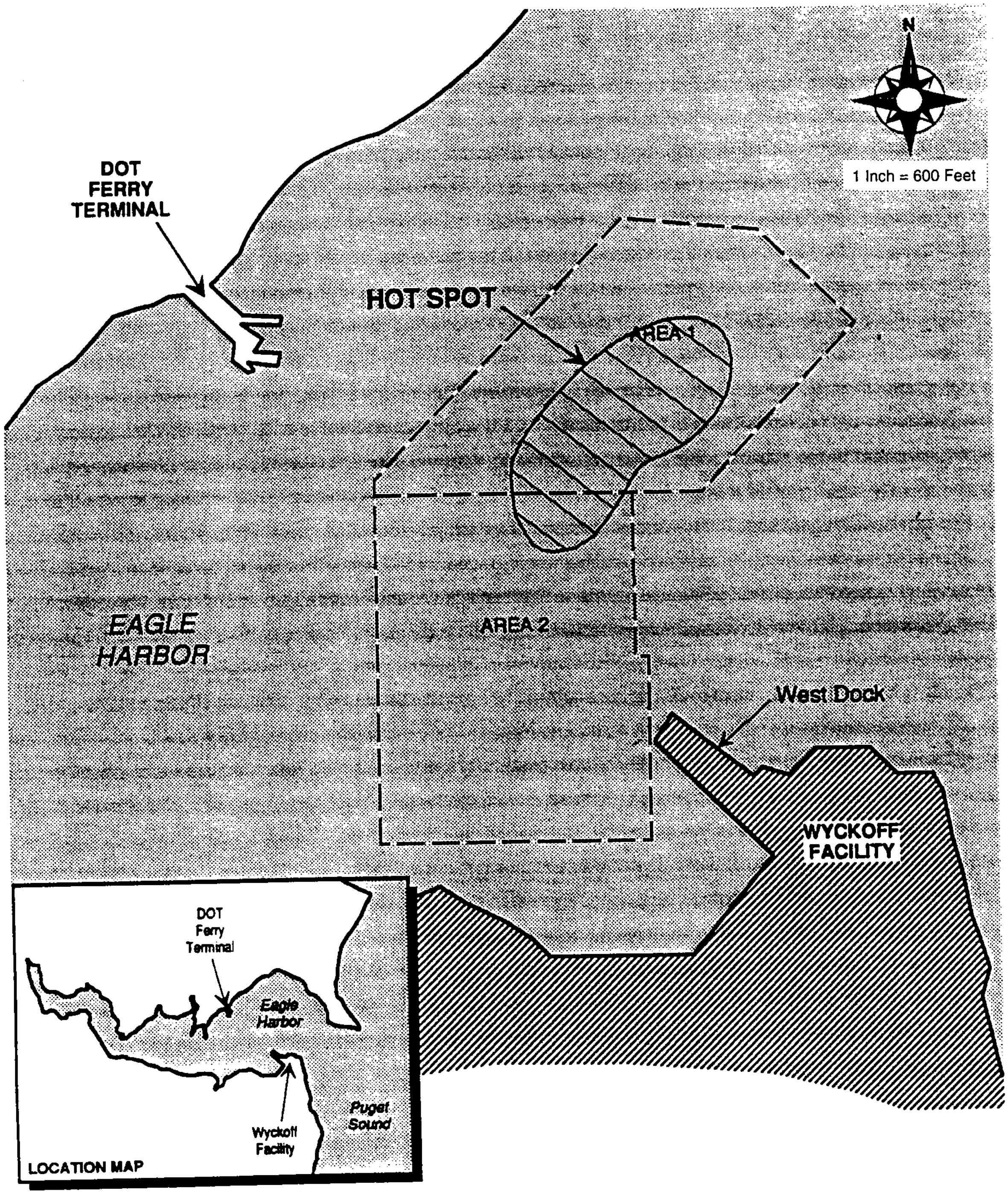

FIGURE 5B-15. Approximate location of PAH-enriched hot spot, Wyckoff/Eagle Harbor Superfund site, East Harbor operable unit.

To ensure lateral continuity and uniform thickness of the cap, accurate navigation of the barges was critical. A global positioning, precision navigation system was used to meet this specification.

Evaluation of Restoration Efforts

The success of capping projects is evaluated by three criteria: adaptive management, ecosystem-level planning, and fail-safe success. These criteria are described below.

Adaptive Management—Special operational management concerns were related to avoiding conflict with the Washington State Ferry that runs through the capping site and concern for the health and safety of operational personnel and the adjacent Eagle Harbor community. Compliance required that the capping material meet thickness and spatial coverage specifications and that cap placement activities not result in increased hazards to the environment. The method of "gentle" placement of a thick sand cap over 52 acres of contaminated muds is far from routine. Therefore, a monitoring program was designed to assess conformance to project objectives while the capping project was underway (U.S. EPA et al. 1994). Sediment traps were placed around the capping area and resuspended sediment recovered from the traps was analyzed for PAHs and metals. Divers monitored sedimentation within adjacent eel grass beds. Bathymetric changes were documented, and sedimentation was also monitored using settlement plates on the bottom as well as sediment profile imaging using a sediment profile camera. Boxcore and grab samples were collected to assess impacts on benthic invertebrates.

A contingency plan was put in place before the project was started to deal with operational problems that could arise regarding critical habitat impacts (Corps and SAIC 1993). Because this project was experimental in nature, it was necessary to have the operation sufficiently flexible to adapt to information gathered from the ongoing compliance monitoring. Special long-term management concerns were focused on Area 1 adjacent to the ferry terminal where propeller wash was known to erode bottom sediments.

Ecosystem-Level Planning—Placement of the Wyckoff/Eagle Harbor project on EPA's National Priorities List underlines the concern for potential far-field effects of the historical contaminant inventories within the harbor. The potential for far-field dispersion of contaminants via the food chain and bottom sediment resuspension was of paramount concern. Because of this concern, all aspects of the capping operation (i.e., "gentle" cap placement along with water and sediment quality monitoring) were designed to minimize far-field impacts. In addition, although bioturbation depths within the harbor were determined to be approximately 10 cm, a conservative management approach was taken to create a cap 100-cm thick. A long-term physical, chemical, and biological monitoring plan is in place to reevaluate cap efficiency.

Safe-Fail Success—Because the cap was put in place in 1994, the long-term success of this habitat restoration project is still to be evaluated. To date, the cap has met all of the design criteria (SAIC 1996a). The area adjacent to the ferry terminal is experiencing some erosion, as expected. Unstable areas of the cap will need close monitoring and may require special management (i.e., recapping) in the future. Using Washington State Sediment Management Standards criteria, the biologically active zone (upper 10 cm) of the sediment cap remains uncontaminated by bioturbational mixing, and there is no evidence of upward diffusion of contaminants from depth. There is limited evidence that the chemistry of the upper 2 cm of the cap (and sediment recovered from deployed sediment traps) is being affected by offsite contamination sources within the harbor. Offsite (ambient) sediments remain uncontaminated relative to Washington State Standards. The sand cap is experiencing normal benthic recolonization, except at one location on the thin margin of the cap where community development appears to be depressed. Continued long-term monitoring of this site will be required to address and remediate localized problem areas as they develop. However, the operational protocol for "gentle" cap emplacement appears to have been successful and may prove valid for similar subtidal habitat restoration projects.

For more information on this project, contact:

U.S. Army Corps of Engineers, Seattle District
4735 East Marginal Way South
Seattle, Washington 98124

Oyster Reef Restoration within Slaughter Creek

Slaughter Creek is located in Dorchester County, Maryland. Slaughter Creek is a tributary of the Little Choptank River in Maryland and represents a part of the Chesapeake Bay estuary (Figure 5B-16). The bottom of the bay in this area is characterized by natural oyster bars. Many of these bars are no longer productive because overharvesting has lowered their profile and altered their substrate so that they are unsuitable for oyster spat recruitment or growth. A restoration plan was developed that included the construction of a dredged material mound on the bay

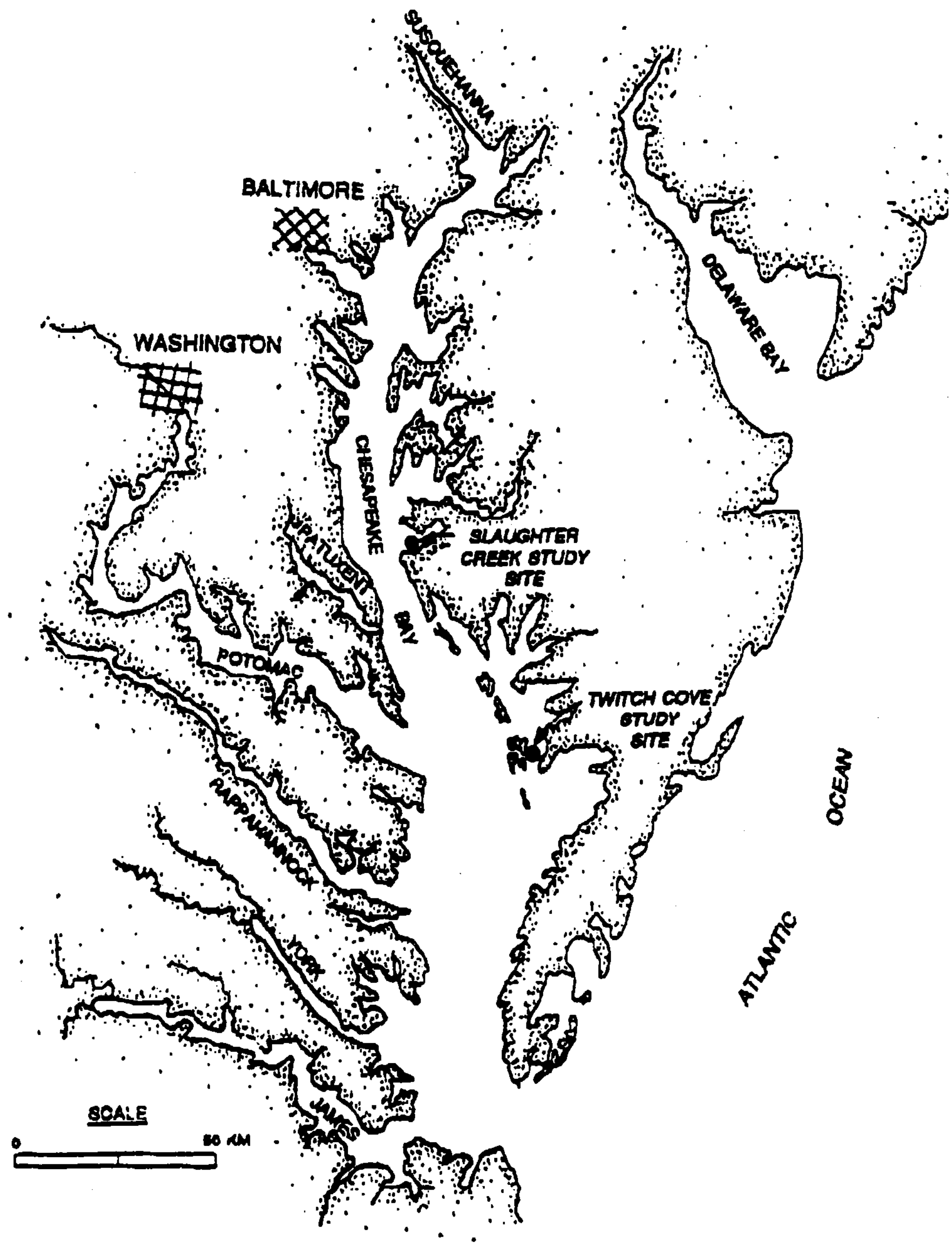

FIGURE 5B-16. Locations of the study sites in the Chesapeake Bay.

bottom. The objectives of the project were to dispose of dredged material and construct a substrate that would be resistant to sedimentation.

Restoration Approach

During the 1987 maintenance dredging of the Slaughter Creek Federal Navigation Channel, approximately 10,700 yd^3 of sediment (60 percent sand and 40 percent silt) were deposited on the bay bottom, thereby creating a 0.9–m–high mound over 3.2 ha. Sediments were dredged using a hydraulic cutterhead pipeline dredge and deposited using a diffuser at the terminal end of the pipeline. The mound was capped with a 20-cm thick layer of oyster shell cultch by washing the shell off barges using high pressure hoses. Spat recruitment was monitored in 1988 and 1989, and the restored oyster bar was compared to two natural bars in the vicinity of the project.

Evaluation of Restoration Efforts

This restoration project was designed to alter substrate conditions to create a more suitable habitat for larval oysters and to reduce the rate of fine-grained sediment deposition over the altered substrate by raising the substrate elevation above the surrounding bay bottom. The success of the project was dependent on the achievement of the design objectives and the validity of the restoration design assumption that substrate-related functions were limiting oyster spat settlement.

Surveys conducted during the monitoring program in 1988, 1989, and 1992 concluded that the restored oyster bar was functioning as well as natural oyster bars in the vicinity of the project (Table 5B-6).

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TABLE 5B-6. SUMMARY DATA FOR SPAT AND OYSTER SIZE AND SURVIVAL AT THE SLAUGHTER CREEK EXPERIMENTAL SHELL CAP AND CASSON AND SUSQUEHANNA NATURAL OYSTER BAR SITES IN 1988, 1989, AND 1990

	Slaughter Creek Experimental Site			Casson Natural Bar			Susquehanna Natural Bar		
	1988	1989	1990	1988	1989	1990	1988	1989	1990
No. Shells in Sample	632	1244	1112	443	410	220	435	334	551
Mean Cultch Size (mm)	47.2	44.8	48.7	67.5	65.1	70.5	79.2	72.6	83.8
Total Cultch Weight (kg)	6.500	11.930	15.303	9.983	15.708	4.900	12.567	10.509	12.466
No. Spat in Sample	41	1	178	52	4	23	60	1	61
Mean Spat Size (mm)	16.5	16.9	14.4	15.1	16.5	15.6	14.5	5.6	16.0
Spat/100 Shells	6.71	0.09	16.01	11.34	0.86	10.46	13.70	0.27	11.07
Spat/kg Cultch	8.00	0.08	11.63	5.21	0.25	4.69	4.78	0.10	4.89
No. Sublegal Oysters in Sample	28	81	88	28	52	11	16	22	17
Mean Sublegal Oyster Size (mm)	37.8	41.7	53.3	46.6	57.2	55.6	53.1	49.2	47.7
Sublegal Oysters/100 Shells	5.97	6.58	7.91	6.06	12.72	5.00	3.65	6.51	3.08
Sublegal Oysters/kg Cultch	4.46	6.79	5.75	2.81	3.31	2.24	1.27	2.09	1.36
No. Legal Oysters in Sample	--	--	2	12	28	2	6	10	3
Mean Legal Oyster Size (mm)	--	--	89.2	87.8	86.4	85.6	90.9	87.9	80.4
Legal Oysters/100 Shells	--	--	0.18	2.79	6.60	0.90	1.50	2.88	0.55
Legal Oysters/kg Cultch	--	--	0.13	1.20	1.78	0.41	0.48	0.95	0.24

Note: Cultch size measured as height (umbo to lip) in mm of unfragmented shell samples from each site.

REFERENCES

Alevizon, W.S., and J.C. Gorham. 1989. Effects of artificial reef deployment on nearby resident fishes. Bull. Mar. Sci. 44(2):646–661.

Aller, R.C. 1982. The effects of macrobenthos on chemical properties of marine sediment and overlying water. pp. 53–102. In: Animal-Sediment Relations: The Biogenic Alteration of Sediments. Volume 2: Topics in Geobiology. P.L. McCall and M.J.S. Tevesz (eds). Plenum Press, New York, NY.

Ambrose, R.F. 1994. Resource replacement alternatives involving constructed reefs in southern California. Agoura Hills, CA.

Armstrong, D.A., B.G. Stevens, and J.C. Hoeman. 1991. Construction dredging impacts on Dungeness crabs in Gray's Harbor, Washington and mitigation of losses by development of intertidal shell habitat. Fisheries Research Institute Report No. FRI-UW-9110. University of Washington, Seattle, WA.

Bahr, L.B., and W.P. Lanier. 1981. The ecology of intertidal oyster reefs of the south Atlantic Coast: a community profile. FWS/OBS-81/15. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC.

Baretta, J., and P. Ruardij (eds). 1988. Tidal flat estuaries: simulation and analysis of the Ems Estuary (Netherlands). Ecological Studies 71. Springer-Verlag, The Netherlands.

Barnes, R.S.K. 1974. Estuarine biology. Edward Arnold, London, England.

Becker, D.S., and K.K. Chew. 1987. Predation on *Capitella* spp. by small-mouthed pleuronectids in Puget Sound, Washington. Fish. Bull. 85:471–479.

Bohnsack, J.A., D.L. Johnson, and R.F. Ambrose. 1991. Ecology of artificial reef habitats and fishes. In: Artificial Habitats for Marine and Freshwater Fisheries. W. Seaman and L.M. Sprague (eds). Academic Press Inc., San Diego, CA.

Bortone, S.A., and J.J. Kimmel. 1991. Environmental assessment and monitoring of artificial habitats. In: Artificial Habitats for Marine and Freshwater Fisheries. W. Seaman and L.M. Sprague (eds). Academic Press Inc., San Diego, CA.

Brumsack, H.J. 1991. Inorganic geochemistry of the German 'Posidonia Shale': palaeoenvironmental consequences. pp. 353–362. In: Modern and Ancient Continental Shelf Anoxia. Special Publication 58. R.V. Tyson, and T.H. Pearson (eds). Geological Society of London, London, England.

Buckley, R.M. 1989. Habitat alterations as a basis for enhancing marine fisheries. Report of the California Cooperative in Oceanic Fisheries Investigations 30:40–45.

Carleton, H.R., I.W. Duedall, P.M.J. Woodhead, and J.H. Parker. 1982. Coal combustion wastes as material for artificial reef construction. pp. 1010–1015. In: Marine Pollution Papers, OCEANS '82, reprinted from the Marine Pollution Sessions of OCEANS '82 Conference Proceedings by the National Oceanic and Atmospheric Administration's Office of Marine Pollution Assessment, Rockville, MD 20852, with permission from The Marine Technology Society and the Institute of Electrical and Electronics Engineers Council on Ocean Engineering.

Carriker, M.R. 1967. Ecology of estuarine benthic invertebrates: a perspective. pp. 442–487. In: Estuaries. Volume 83. G.H. Lauff (ed). American Association for the Advancement of Science, Washington, DC.

Christian, R.R., and R.L. Wetzel. 1978. Interaction between substrate, microbes, and consumers of *Spartina* detritus in estuaries. pp. 93–114. In: Estuarine Interactions. M.L. Wiley (ed). Academic Press, New York, NY.

Clarke, D., and R. Kasul. 1994. Habitat value of offshore dredged material berms for fishery resources. pp. 938–945. In: Dredging '94, Proceedings of the Second International Conference on Dredging and Dredged Material Placement. E.C. McNair, Jr. (ed). American Society of Civil Engineers, New York, NY.

Collins, K.J., E.K. Free, A.C. Jensen, and S. Thompson. 1993. Analysis of Poole Bay, U.K., lobster data. In: International Council for the Exploration of the Sea, 1993 Council Meeting Papers. ICES, Copenhagen, Denmark.

Corps. 1989. Final environmental impact statement. Gray's Harbor Navigation Improvement Project. Supplement I. U.S. Army Corps of Engineers, Seattle District, Seattle, WA.

Corps and SAIC. 1993. Final removal action management plan, East Harbor Operational Unit of the Wyckoff/Eagle Harbor Superfund site, Bainbridge Island, Washington. U.S. Army Corps of Engineers, Seattle, WA, and Science Applications International Corporation, Bothell, WA.

Davis, G.E. 1985. Artificial structures to mitigate marina construction impacts on spiny lobster, *Panulirus argus*. Bull. Mar. Sci. 37:151–156.

Day, J.W., C.A.S. Hall, W.M. Kemp, and A. Yanez-Aranciba. 1989. Estuarine Ecology. John Wiley & Sons, New York, NY.

Duedall, I.W., and M.A. Champ. 1991. Artificial reefs: emerging science and technology. Oceanus 34(1):94–101.

Duggins, D., C.A. Simenstad, and J.A. Estes. 1989. Magnification of secondary production by kelp detritus in coastal marine ecosystems. Science 245:170–174.

Emery, K.O. 1967. Estuaries and lagoons in relation to continental shelves. pp. 9–11. In: Estuaries. Volume 83. G.H. Lauff (ed). American Association for the Advancement of Science, Washington, DC.

Fairbridge, R. 1980. The estuary: its definition and geodynamic cycle. pp. 1–35. In: Chemistry and Biochemistry of Estuaries. E. Olausson and I. Cato (eds). John Wiley & Sons, New York, NY.

Feigenbaum, D., M. Bushing, J. Woodward, and A. Friedlander. 1989. Artificial reefs in Chesapeake Bay and nearby coastal waters. Bull. Mar. Sci. 44:734–742.

Fitzhardinge, R.C., and J.H. Bailey-Brock. 1989. Colonization of artificial reef materials by corals and other sessile organisms. Bull. Mar. Sci. 44(2):567–579.

Forbes, T.L., V.E. Forbes, and M.H. Depledge. 1994. Individual physiological responses to environmental hypoxia and organic enrichment: implications for early soft-bottom community succession. J. Mar. Res. 52:1081–1100.

Foster, M.S., and D.R. Schiel. 1985. The ecology of giant kelp forests in California: a community profile. U.S. Fish and Wildlife Service Biological Report 85(7.2).

Gerlach, S.A. 1971. On the importance of marine meiofauna for benthos communities. Oecologia 6:176–190.

Germano, J.D. 1983. Infaunal succession in Long Island Sound: animal-sediment interactions and the effects of predation. Dissertation. Yale University, Biology Department, New Haven, CT.

Germano, J.D., D.C. Rhoads, and J.D. Lunz. 1994. An integrated, tiered approach to monitoring and management of dredged material disposal sites in the New England Region. Special Technical Report, Contribution 87. U.S. Army Corps of Engineers, New England Division, Waltham, MA.

Groen, P. 1969. Physical hydrology of coastal lagoons. pp. 275–280. In: Coastal Lagoons, a Symposium. A.A. Castañares and F.B. Phleger (eds). Universidad Nacional Autonoma de Mexico, Ciudad Universitaria, Mexico, UNAM/UNESCO.

Hay, M.E., and J.P. Sutherland. 1988. The ecology of rubble structure of the south Atlantic bight: a community profile. U.S. Fish and Wildlife Service Biological Report 85(7.20).

Hayes, M.O. 1978. Impact of hurricanes on sedimentation in estuaries, bays, and lagoons. pp. 323–343. In: Estuarine Interactions. M. Wiley (ed). Academic Press, New York, NY.

Hayes et al. 1988. Demonstration of innovative and conventional dredging equipment at Calumet Harbor, Illinois. Miscellaneous Paper EL-88-1. U.S. Army Corps of Engineers, Waterways Experimental Station, Vicksburg, MS.

Hueckel, G.J., R.M. Buckley, and B.L. Benson. 1989. Mitigating rocky habitat loss using artificial reefs. Bull. Mar. Sci. 44(2):913–922.

Johnson, R.G. 1972. Conceptual models of benthic marine communities. pp. 148–159. In: Models in Paleobiology. T.J.M. Schopf (ed). Freeman, Cooper & Co., San Francisco, CA.

Jones and Stokes. 1991. Post-project monitoring at Slip 5 mitigation area, 1988 and 1989. Report prepared for the Port of Tacoma, WA. Jones and Stokes Associates, Inc., Seattle, WA.

Kennish, M.J. 1990. Ecology of estuaries. Volume II: Biological Aspects. CRC Press, Boston, MA.

Kranz, P. 1974. The anastrophic burial of bivalves and its paleontological significance. J. Geol. 82:237–265.

Levinton, J.S. 1972. Stability and trophic structure in deposit-feeding and suspension-feeding communities. Am. Nat. 106:472–486.

Levinton, J.S., T.S. Bianchi, and S. Stewart. 1984. What is the role of particulate organic matter in benthic invertebrate nutrition? Bull. Mar. Sci. 35:270–282.

Lucy, J.A., and C.G. Barr. 1989. Development and implementation of a catch and effort data collection system for monitoring trends in fishing success on Virginia's artificial fishing reefs, 1987–1989. NTIS PB90-168188/GAR.

Lynch, M.P., B.L. Laird, N.B. Theberge, and J.C. Jones (eds). 1976. An assessment of estuarine and nearshore marine environments. Special Report in Applied Marine Science and Ocean Engineering No. 93, Virginia Institute of Marine Science, Gloucester Point, VA.

Martens, C.S. 1984. Recycling of organic carbon near the sediment-water interface in coastal environments. Bull. Mar. Sci. 35:566–575.

May, B., D. Pabst, and S. McDowell. 1994. Dioxin capping management and monitoring program: design and implementation. pp. 1027–1036. In: Dredging '94, Proceedings of the Second International Conference on Dredging and Dredged Material Placement. E.C. McNair, Jr. (ed). American Society of Civil Engineers, New York, NY.

McGurrin, J.M, R.B. Stone, and R.J. Sousa. 1989. Profiling United States artificial reef deployment. Bull. Mar. Sci. 44:1004–1013.

MEC. 1993. Deep water mitigation alternatives for port development. Report submitted to the California Association of Port Authorities, Sacramento, CA. Marine Environmental Consultants, Carlsbad, CA.

Merkel, K. 1995. Personal communication (telephone conversation with T. Thompson, SAIC, Bothell, WA, regarding eelgrass migration). Merkel and Associates, Spring Valley, CA.

Merkel, K., and R. Hoffman. 1990. The use of dredge materials in the restoration of eelgrass meadows: a southern California perspective. Proceedings of a Regional Workshop: Beneficial Uses of Dredged Material in the Western United States, May 21–25, 1990. San Diego, CA. M.S. Landin (ed). U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Merrill, J. 1995. Personal communication (telephone conversation with T. Thompson, SAIC, Bothell, WA). Michigan State University, Lansing, MI.

Merrill, J.E., and D.M. Gillingham. 1991. Bull kelp cultivation handbook. Publication No. NCRI-T-91-011. National Coastal Resource Institute, Newport, OR.

Meyers, M.B., E.N. Powell, and H. Fossing. 1987. Microdistribution of interstitial meiofauna, oxygen, and sulfide gradients, and the tubes of macro-infauna. Mar. Ecol. Prog. Ser. 35:223–241.

Montague, C.L., S.M. Bunker, E.B. Haines, M.L. Pace, and R.L. Wetzel. 1981. The ecology of a salt marsh. Volume 38: Ecological Studies. L.R. Pomeroy and R. Wiegert (eds). Springer-Verlag, New York, NY.

Mueller, A. J., and E.D. Whitehead. 1986. Freeport Marsh development and wildlife habitat mitigation analysis. U.S. Fish and Wildlife Service, Clear Lake Ecological Services Field Office, Houston, TX.

Murray, P. 1996. Sediment core chemistry data summary from the Mill-Quinnipiac River mound. Contribution 103. U.S. Army Corps of Engineers, New England Division, Waltham, MA.

Murray, P., D. Carey, and T. Fredette. 1994. Chemical flux of pore water through sediment caps. pp. 1008–1016. In: Dredging '94, Proceedings of the Second International Conference on Dredging and Dredged Material Placement. E.C. McNair, Jr. (ed). American Society of Civil Engineers, New York, NY.

Nelson, C.H., and K.R. Johnson. 1987. Whales and walruses as tillers of the seafloor. Sci. Amer. 256(2):112–117.

Nelson, E.E., A.L. Vanderheiden, and A.D. Schmidt. 1994. Eagle Harbor Superfund project. pp. 1122–1131. In: Dredging '94, Proceedings of the Second International Conference on Dredging and Dredged Material Placement. E.C. McNair, Jr. (ed). American Society of Civil Engineers, New York, NY.

NOAA. 1990. Estuarine habitat program implementation plan. Coastal Ocean Program, FY 1990. National Oceanic and Atmospheric Administration, Washington, DC.

NRC. 1990. Managing troubled waters: the role of marine environmental monitoring. National Research Council, Marine Board, Washington, DC. National Academy Press, Washington, DC.

NRC. 1994. Restoring and protecting marine habitat: the role of engineering and technology. National Research Council, Marine Board, Washington, DC. National Academy Press, Washington, DC.

Odum, E.P. 1969. The strategy of ecosystem development. Science 16:262–270.

Odum, H.T., B.J. Copeland, and E.A. McMahan. 1974. Coastal ecological systems of the United States. Conservation Foundation, Washington, DC.

Palermo, M.R., C.R. Lee, and N.R. Francingues. 1989. Management strategies for disposal of contaminated sediments. pp. 200–220. In: Contaminated Marine Sediments—Assessment and Remediation. National Research Council, Commission on Engineering and Technical Systems, Committee on Contaminated Marine Sediments, Marine Board. National Academy Press, Washington, DC. 493 pp.

Pearson, T.H., and R. Rosenberg. 1978. A comparative study of the effects on the marine environment of wastes from cellulose industries in Scotland and Sweden. Ambio 5:77–79.

Pier 400 Design Consultants. 1995. Pier 400 dredging and landfill project. Post disposal survey permanent shallow water habitat. Prepared for the Port of Los Angeles, San Pedro, CA.

Polovina, J.J. 1989. Artificial reefs: nothing more than benthic fish aggregators. Report of the California Cooperative in Oceanic Fisheries Investigations 30:37–39.

Postma, H. 1969. Chemistry of coastal lagoons. pp. 421–430. In: Coastal Lagoons, A Symposium. A.A. Castañares and F.B. Phleger (eds). Universidad Nacional Autonoma de Mexico, Mexico City, Mexico.

Pritchard, D. 1967. Observations of circulation in coastal plain estuaries. pp. 37–44. In: Estuaries. Volume 83. G.H. Lauff (ed). American Association for the Advancement of Science, Washington, DC.

Reice, S.R. 1994. Nonequilibrium determinants of biological community structure. Am. Sci. 82:424–435.

Rhoads, D.C., and L.F. Boyer. 1982. The effects of marine benthos on physical properties of sediments: a successional perspective. pp. 3–52. In: Animal-Sediment Relations. Topics in Geobiology Series. Volume 2. P.L. McCall and M.J.S. Tevesz (eds). Plenum Press, New York, NY.

Rhoads, D.C., and J.D. Germano. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142:291–308.

Rhoads, D.C., and D.K. Young. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. J. Mar. Research 28:150-178.

Rhoads, D.C., and D.K. Young. 1971. Animal-sediment relations in Cape Cod Bay, Massachusetts. Reworking by *Molpadia oolitica* (Holothuroidea). Mar. Biol. 11:255-261.

Rhoads, D.C., P.L. McCall, and J.Y. Yingst. 1978. Disturbance and production on the estuarine seafloor. Am. Sci. 66:577-586.

Rice, D.L., and D.C. Rhoads. 1984. Early diagenesis of organic matter and the nutritional value of sediment. pp. 59-97. In: Ecology of Marine Deposit Feeders. G. Lopez, G. Taghon, and J. Levinton (eds). Lecture Notes on Coastal and Estuarine Studies. Springer-Verlag, New York, NY.

Roman, M.R., and K.R. Tenore. 1984. Detritus dynamics in aquatic ecosystems: an overview. Bull. Mar. Sci. 35:257-260.

Rothschild Ault Group. 1992. Review of coupled differential equation ecosystem models. Unpublished report dated November 1992. Solomons Island, MD.

Ryther, J.H. 1969. Photosynthesis and fish production in the sea. Science 166:72-76.

SAIC. 1987. Environmental information in support of site designation documents for the Foul Area disposal Site. SAIC Report No. SAIC-85/7528&93. Submitted to the U.S. Army Corps of Engineers, New England Division, Waltham, MA. Science Applications International Corporation, Bothell, WA.

SAIC. 1994. Observation recorded during a water sampling event on Brakes Bayou, Beamont, TX. October 1994. Science Applications International Corporation, Bothell, WA.

SAIC. 1995. Sediment capping of subaqueous dredged material disposal mounds: an overview of the New England Experience 1979-1993. Special Technical Report, Contribution 95. U.S. Army Corps of Engineers, New England Division, Waltham, MA.

SAIC. 1996a. 1995 environmental monitoring report, long term monitoring program Wycoff/Eagle Harbor Superfund site East Harbor Operational Unit Bainbridge Island, Washington (Final Report). U.S. Environmental Protection Agency Region 10, Seattle, WA; U.S. Army Corps of Engineers, Seattle, WA; and Science Applications International Corporation, Bothell, WA.

SAIC. 1996b. The use of ecosystem models in planning environmental restoration projects. Prepared for U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS. Science Applications International Corporation, Bothell, WA.

Schubel, J.R. 1971. Some notes on turbidity maxima. pp. VIII-1-28. In: The Estuarine Environment: Estuaries and Estuarine Sedimentation. Short Course Lecture Notes, Wye Institute, J. Schubel (Convenor). M.O. Hayes, D.W. Pritchard, and J.R. Schubel (Lecturers). American Geological Institute, Falls Church, VA.

Scott, K.J. 1989. Effects of contaminated sediments on marine benthic biota and communities. pp. 132-154. In: Contaminated Marine Sediments—Assessment and Remediation. National Academy Press, Washington, DC.

Scott, A.J., R.A. Hoover, and J.H. McGown. 1969. Effects of hurricane "Beulah," 1967 on Texas coastal lagoons and barriers. pp. 221-236. In: Coastal Lagoons, A Symposium. A.A. Castañares, and F.B. Phleger (eds). Universidata Nacional Autonoma de Mexico, Mexico City.

Seaman, W., and L.M. Sprague. 1991. Artificial habitats for marine and freshwater fisheries. Academic Press, Inc., San Diego, CA.

Seaman, W., R.M. Buckley, and J.J. Polovina. 1989. Advances in knowledge and priorities for research, technology, and management related to artificial aquatic habitats. Bull. Mar. Sci. 44(2):527-532.

Sheehan, P.J. 1984. Effects on individuals and populations. pp. 23-50. In: Effects of Pollutants at the Ecosystem Level. P.J. Sheehan, D.R. Miller, G.C. Butler, and P. Bourdeau (eds). John Wiley & Sons, New York, NY.

Sheehy, D.J., and S.F. Vik. 1992. Developing prefabricated reefs: an ecological and engineering approach. pp. 542-582. In: Restoring the Nation's Marine Environment. G.W. Thayer (ed). Maryland Sea Grant, College Park, MD.

Simenstad, C.A., C.D. Tanner, R.M. Thom, and L.L. Conquest. 1991. Estuarine habitat assessment protocol. EPA 910/9-91-037. U.S. Environmental Protection Agency Region 10, Puget Sound Estuary Program, Seattle, WA.

Stivers, C.E., and R. Sullivan. 1994. Restoration and capping of contaminated sediments. pp. 1017-1026. In: Dredging '94, Proceedings of the Second International Conference on Dredging and Dredged Material Placement. E.C. McNair, Jr. (ed). American Society of Civil Engineers, New York, NY.

Tenore, K.R., R.B. Hanson, J. McClain, A.E. Maccubbin, and R.E. Hodson. 1984. Changes in composition and nutritional value to a benthic deposit feeder of decomposing detritus pools. Bull Mar. Sci. 35:299–311.

Thompson, T. 1982. Estuarine distribution of the rockweed, *Fucus distichus* L. Thesis. University of British Columbia, Canada.

Tyson, R.V., and T.H. Pearson. 1991. Modern and ancient continental shelf anoxia: an overview. pp. 1–24. In: Modern and Ancient Continental Shelf Anoxia. Special Publication 58. R.V. Tyson and T.H. Pearson (eds). Geologic Society of London, London, England.

U.S. EPA. 1990. Environmental monitoring and assessment program: ecological indicators. Appendix A: indicator fact sheets for near-coastal waters. C. Hunsaker and D.E. Carpenter (eds). U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

U.S. EPA, Corps, and SAIC. 1994. On scene coordinator's report, statement of findings, East Harbor Operational Unit removal action Wycoff/Eagle Harbor Superfund site Bainbridge Island, Washington (Final Report). U.S. Environmental Protection Agency Region 10, Seattle, WA; U.S. Army Corps of Engineers, Seattle, WA; and Science Applications International Corporation, Bothell, WA.

USFWS. 1985. Habitat suitability index models: great blue herons. Biological Report 82(10.99). U.S. Fish and Wildlife Service, Washington, DC.

Valente, R.M., D.C. Rhoads, J.D. Germano, and V.J. Cabelli. 1992. Mapping of benthic enrichment patterns in Narragansett Bay, Rhode Island. Estuaries 15:1–17.

Valiela, I., J. Wilson, R. Buchsbaum, C. Rietsma, D. Bryant, K. Foreman, and J. Teal. 1984. Importance of chemical composition of salt marsh litter on decay rates and feeding by detritivores. Bull. Mar. Sci. 35:261–269.

Wiley, M.B. 1994. DAMOS capping model verification. Special Technical Report, Contribution 89. U.S. Army Corps of Engineers, New England Division, Waltham, MA.

Yingst, J.Y., and D.C. Rhoads. 1980. The role of bioturbation in the enhancement of bacterial growth rates in marine sediments. pp 407–421. In: Marine Benthic Dynamics. K.R. Tenore and B.C. Coull (eds). University of South Carolina Press, Columbia, SC.

Young, D.K. 1971. Effects of infauna on the sediment and seston of a subtidal environment. Vie et Milieu Supp. 22:557–571.

5C. ESTUARINE AND COASTAL WETLANDS

Barry Vittor and David Yozzo

Estuarine and coastal wetlands are plant-based communities located at the interface between rivers and the ocean that are periodically flooded by salt, brackish, or freshwater. These communities are found mostly in areas of low wave energy, near or in the intertidal zone, along bays and rivers. Physical factors are highly variable in these environments, especially water salinities and the depth and duration of tidal inundation. Estuarine and coastal wetlands help prevent coastal erosion, provide an area to store flood waters, maintain and enhance water quality, and provide refuge for aquatic organisms, including many commercially important species.

In all estuarine and coastal wetland ecosystems, the spatial complexity provided by the presence of vegetation attracts animals that inhabit these areas. The botanical component of estuarine and coastal wetlands provides a source of food (through detrital materials and algal epiphytes), refuge from predation, and a breeding site. In seagrass ecosystems, for example, population densities of benthic and epibenthic invertebrates are much higher in grassbeds than in nearby unvegetated areas (Lewis and Stoner 1983; Lewis 1984). Maintenance of the rich biological communities in estuarine and coastal wetlands is therefore dependent on the viability of plant communities, which are dependent on sensitive and narrowly defined environmental factors.

ECOSYSTEM PROFILE

The effects of salinity and the range of tidal flooding in estuarine and coastal wetlands are largely responsible for the delineation of distinct zones of vegetation types. Most of these ecosystems are populated by only a few plant species, and plant diversity tends to increase from the edge of the water toward higher elevations. The faunal communities found in these environments are often a function of the type of plants that form the basis of, or define, the ecosystem. Four distinct estuarine and coastal wetland ecosystems are of significant concern when developing restoration projects: salt marshes, freshwater tidal wetlands, mangrove forests, and seagrass beds.

Salt Marshes

Salt marsh vegetation is dominated by grasses (Poaceae), rushes (Juncaceae), or a combination of these two families. The extreme variability in environmental factors, such as nutrient availability (particularly nitrogen), duration and depth of tidal inundation, and pore water salinity limits plant species in salt marshes. The spatial extent of the major zones of vegetation is largely determined by elevation and the resultant effect on the tidal flooding regime.

Geographic Distribution

Salt marshes are found in all coastal areas of North America, except the southern half of Florida and certain areas of Texas, where they are replaced by mangrove forests. Salt marshes that occur along the Pacific coast are far less extensive and less uniform than those found on the Atlantic and Gulf coasts.

Zonation Within Habitats

Woodhouse and Knutson (1982) divided East coast marshes into three general types: New England, mid-Atlantic, and south Atlantic. The dominant plant in the intertidal zone on the Atlantic coast of the United States is smooth cordgrass (*Spartina alterniflora*). This species generally occurs between mean high water and mean low water and exhibits considerable variation in growth form (i.e., tall, medium, and short), as determined primarily by tidal flooding frequency and duration. Above mean high water, floral composition of salt marshes increases in diversity and varies with latitude.

In New England, smooth cordgrass occurs in a relatively narrow zone along tidal creeks or the marsh edge. The high marsh (above mean high water) is dominated by saltmeadow cordgrass (*S. patens*) and saltgrass (*Distichlis spicata*). Monospecific stands of blackgrass (*Juncus gerardi*) are found as elevation increases. Mid-Atlantic and south Atlantic salt marshes are characterized by the tall form of smooth cordgrass along creekbanks, grading to a wide band of medium to short smooth cordgrass as elevation gradually increases. In the mid-Atlantic, saltmeadow cordgrass covers most of the area above mean high water, with occasional stands of black needlerush (*J. roemerianus*). Along the south Atlantic coast, black needlerush dominates in the high marsh. Unvegetated salt pannes are common intertidal landscape features in Atlantic coast salt marshes, and these pannes may be fringed by halophytes such as glasswort (*Salicornia* spp.) and saltwort (*Baetis maritima*).

Brackish marshes generally occur in the mid to upper reaches of tidal rivers along the Atlantic coast. Depending on the degree of freshwater input and its effect on the local salinity regime, these marshes may be dominated by either smooth cordgrass or big cordgrass (*S. cynosuroides*). The latter appears to predominate when interstitial salinities range from 2–14 ppt (Schubauer and Hopkinson 1984).

Bulrush (*Scirpus americana*) and pickerelweed (*Pontedaria cordata*) may also be present in mixed stands associated with big cordgrass. In the Albemarle-Pamlico Sound region of North Carolina, tidal amplitude is strongly inhibited by the Outer Banks, and the extensive microtidal brackish marshes that form along the mainland are *Juncus*-dominated systems.

Salt marshes of the northeastern Gulf of Mexico are also mostly *Juncus*-dominated systems, with up to 92 percent of Mississippi salt marshes comprised of black needlerush (Eleuterius 1976). The lower edge of these marshes contains smooth cordgrass, and at higher elevations above the *Juncus* zone saltmeadow cordgrass and saltgrass occur. A shrub zone usually occurs before the treeline and may contain several species, including marsh elder *Iva frutescens* (Kruczynski 1982).

The intertidal zone along the western Gulf of Mexico (Webb 1982) is dominated by smooth cordgrass. Above mean high water a narrow zone of saltgrass often occurs. A zone of saltmeadow cordgrass dominates the marsh above the saltgrass, although these two plants are often mixed together. Throughout Texas and Louisiana, saltmeadow cordgrass predominates above the saltgrass zone. In Louisiana brackish marshes, stands of big cordgrass may occur in the intertidal zone, especially in areas of slightly higher elevation. South of Corpus Christi, Texas, the vast marshes are replaced by wind-tidal flats (Brown et al. 1976), with little smooth cordgrass occurring, and high marsh areas composed of several species, including saltflat grass (*Monanthochloe littoralis*), saltwort (*Batis maritima*), and glasswort (*Salicornia* spp.) (Webb 1982).

Along central and southern California, the intertidal zone is dominated by Pacific cordgrass (*Spartina foliosa*). Where freshwater has a great influence, Pacific cordgrass is replaced by brackish species (Knutson and Woodhouse 1982), including bulrush (*Scirpus robustus*). The high marsh (above mean high water) is dominated by pickleweed (*Salicornia virginia*), and saltgrass is common.

In the Pacific northwest, where freshwater influence is high, no single species dominates the intertidal zone. Species found here include Lyngbye's sedge (*Carex lyngbyei*), three-square bulrush (*Scirpus californicus*), and spike rushes (*Eleocharis* spp.). In the upper elevations, saltgrass is common (Knutson and Woodhouse 1982).

Biological Community

Although few animals feed directly on live grasses and plants, dead and decaying plant material is a source of nutrition for many species in salt marshes. This material forms the basis of a detrital food chain in and near the marshes (Figures 5C-1 and 5C-2). In the intertidal salt marsh, where cordgrass or needlerush often predominate, most of the plant material is initially colonized and broken down by bacteria and fungi. The breakdown of cellulose makes vegetative protein available to numerous detritivores, including snails, bivalves, shrimps, and insect larvae. Much of the intertidal plant biomass in a salt marsh ecosystem is exported as detritus from the estuary via currents and tidal flushing (de la Cruz 1979). The high marsh (above mean high water) also is productive in terms of plant growth, but unlike lower areas that are regularly flooded, plant material is not exported (Woodhouse and Knutson 1982). Algal epiphytes that grow on the grassblades further contribute to primary production and are grazed upon by species such as the marsh periwinkle (*Littorina* spp.).

Benthic fauna within the grassy marsh are mostly invertebrates, including bivalves (e.g., the ribbed mussel *Guekensia demissa*) and annelid worms, living in or on the soft bottom of the marsh (Kneib 1984b). Small pools and moist sediments permit survival of these species between high tides. Meiofaunal species are diverse, consisting primarily of nematodes, copepods, and polychaete annelids (Bell 1979), which are fed upon by larval fishes and crustaceans such as grass shrimp (*Palaemonetes* spp.). Many benthic invertebrates tend to aggregate in close association with culms of vegetation in these ecosystems (Rader 1984), indicating that the presence of plants such as smooth cordgrass plays a key role in the distribution and abundance patterns of saltmarsh-inhabiting infauna.

Epibenthic macrofauna found within salt marshes include species that can tolerate wide fluctuations in salinity and temperature (e.g., blue crabs [*Callinectes sapidus*]). At higher elevations of the intertidal zone, fiddler crabs (*Uca* spp.) construct burrows in the soft sediments. Many species, such as the mummichog *Fundulus heteroclitus* and the grass shrimp *Palaemonetes pugio*, are resistant to desiccation during periods of low tidal levels (Knieb 1987). Other environmental disturbance factors, such as high rates of freshwater discharge from rivers, will drive many species seaward until ambient salinities rise in the marsh. The composition of salt marsh faunal communities thus tends to fluctuate diurnally and seasonally and also varies within populations (e.g., distribution of age classes).

Species such as spot (*Leiostomus xanthurus*), brown shrimp (*Penaeus aztecus*), and blue crabs inhabit shallow subtidal areas at low tide. During rising tides, they move onto the vegetated marsh surface to feed and seek refuge from predators (Zimmerman and Minello 1984). As with infauna, these species are positively associated with dense vegetation, relative to bare substrata (Zimmerman and Minello 1984; Thomas et al. 1990).

Salt marshes are an important habitat for reproduction and maturation, both for resident species and for transients and offshore spawners that use the marsh

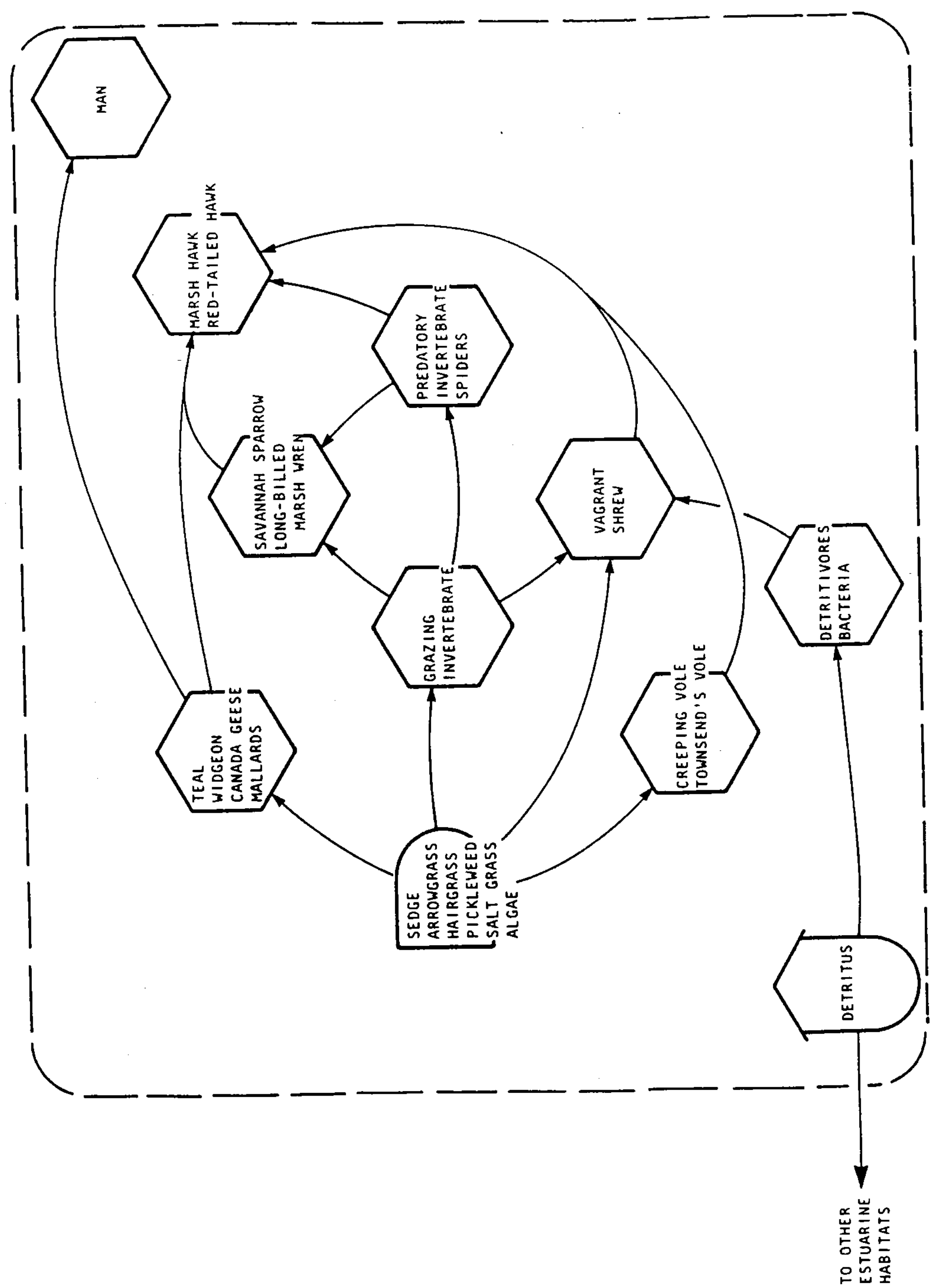
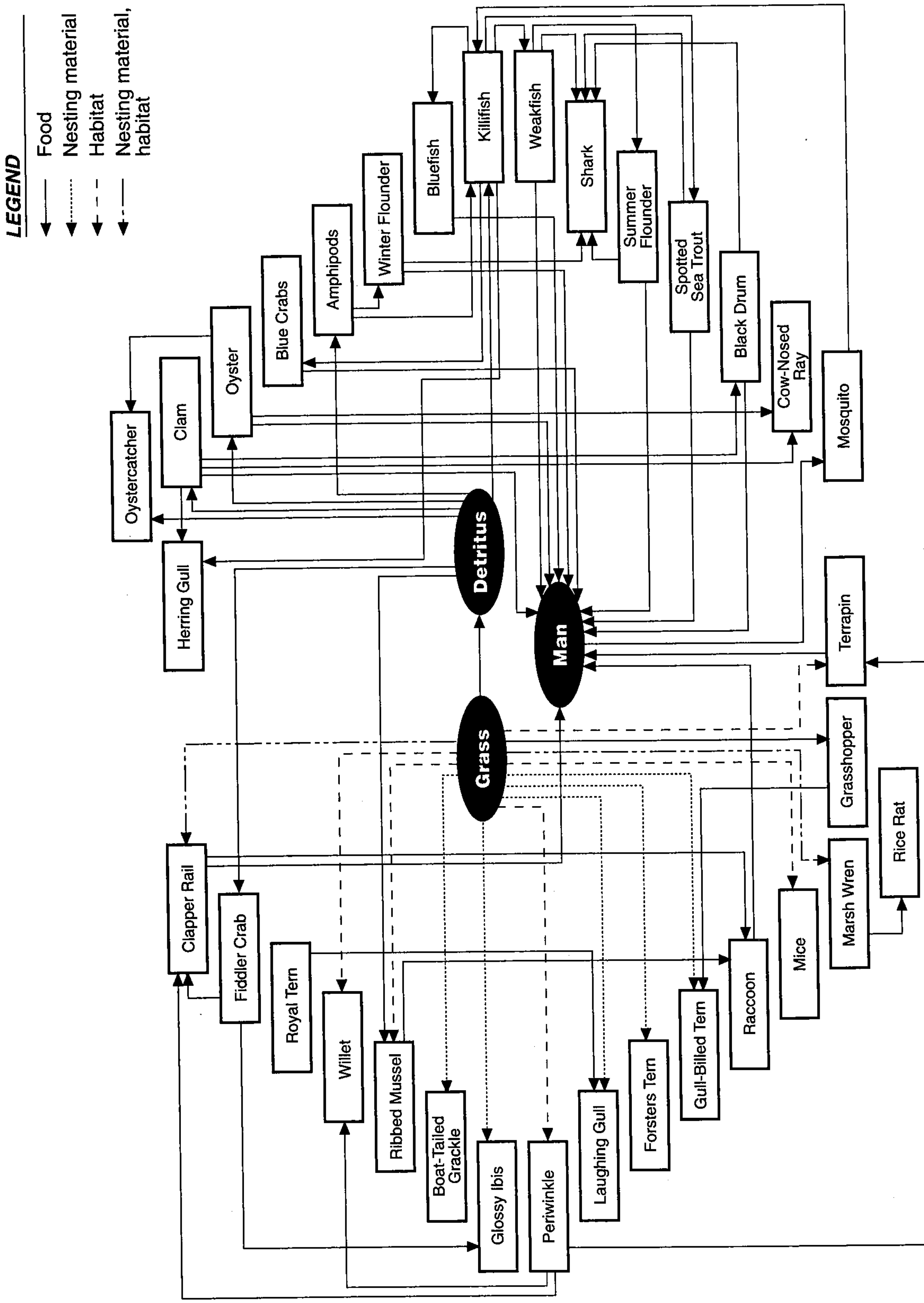


FIGURE 5C-1. Food web for emergent brackish marsh habitat.

Source: Proctor et al. (1980).



Source: Modified from Wass and Wright (1969).

FIGURE 5C-2. Diagram showing the ultimate utilization of smooth cordgrass on the eastern shore of Virginia.

during early life stages. During spring tides, the mummichog deposits eggs on the marsh surface, either attaching them to blades of smooth cordgrass or placing them in empty shells of the ribbed mussed (Taylor and DiMichelle 1983). The vegetated intertidal marsh surface is the primary nursery habitat for several species of killifish in Atlantic and Gulf coast marshes (Kneib 1984a; Rozas and Lasalle 1990). Postlarval and juvenile grass shrimp recruit to the marsh surface during mid to late summer and mature in shallow intertidal habitats (Kneib 1987). Offshore spawners and other transient nekton (including many commercially important species) use the densely vegetated marsh edge as a foraging site and refuge from predation. In the northwestern Gulf of Mexico, juvenile blue crabs show an increased dependence on salt marsh habitat as a nursery, especially in areas where seagrasses are absent or limited in areas (Thomas et al. 1990). Brown shrimp (*Penaeus aztecus*), another offshore spawner, are likewise heavily dependent on salt marsh habitat during early life stages (Zimmerman and Minello 1984).

The irregularly flooded high marsh is used as a nesting area by various avifauna, including clapper rails (*Rallus longirostris*), terns (*Sterna* spp.), and willet (*Catoptrophorus semipalmatus*). Rat snakes (*Elaphe* spp.) and small mammals, including raccoons (*Procyon lotor*), meadow voles (*Microtus pennsylvanicus*), and rice rats (*Orozymys palustris*), nest in the high marsh. the relatively high species diversity of birds observed in high salt marsh habitats has been attributed to the "edge-effect" of the marsh-upland ecotone that is attractive to both shorebirds and arboreal species (Nixon 1982).

Tidal Freshwater Wetlands

Tidal freshwater wetlands are those habitats with low salinities (<0.5 ppt) that are nonetheless subject to tidal fluctuation. Instead of the domination of vertical zones by one or two plant species (e.g., *Spartina*), freshwater intertidal wetlands frequently include mixtures of many species. This results in seasonal variation in macrophyte domination, with annual and perennial plant species alternating both in areal extent and biomass (Odum 1988).

Geographic Distribution

Tidal freshwater wetlands are found in all coastal areas of North America where rivers exhibit moderate to strong tidal influence. They are most extensive along the Atlantic coast between Georgia and New England, especially in the mid-Atlantic/Chesapeake Bay region and along the coastal plain rivers of South Carolina and Georgia. Tidal freshwater wetlands are found in the northeast Gulf of Mexico, particularly in the Mobile Delta in Alabama. Extensive freshwater wetlands can be found in coastal Louisiana; however, these systems are characterized by irregular, low amplitude, wind-driven tides. On the West coast, tidal freshwater wetlands are most extensive in the Pacific Northwest, where freshwater influence and tidal amplitude is greater than in southern California.

Zonation Within Habitats

High species diversity can complicate efforts to delineate distinct zones of vegetation in tidal freshwater wetlands. Many marsh types are classified according to the dominant species found within them (Odum et al. 1984). Spatterdock (*Nuphar lutecum*) communities are found in subtidal, open waters from Massachusetts to northern Florida and exist in pure stands. This species has floating leaves that are attached to stalks rising from embedded rhizomes. Cattail (*Typha* spp.) also forms the basis of distinct communities and is common in all freshwater environments of North America (Tiner 1993). It occurs as stout reeds, from shallow waters up into the high intertidal zone, and is usually found in mixed stands with other species, including arrow-arum (*Peltandra virginica*) and smartweed (*Polygonum densiflorum*). Other tidal freshwater communities that are based on distinct vegetative types (Odum et al. 1984) include arrow-arum/ pickerelweed (*Peltandra virginica/Pontedaria cordata*), wild rice (*Zizania aquatica*), and giant cutgrass (*Zizaniopsis mileacea*).

It is common to observe diverse plant communities in tidal freshwater wetlands. For example, in the eastern United States (Tiner 1993), submerged vegetation communities may include bushy pondweed (*Najas guadalupensis*), milfoil (*Myriophyllum* sp.), wild celery (*Vallisneria americana*), and coontail (*Ceratophyllum demersum*). The intertidal zone may include mixtures of cattail, arrow-arum, smartweed, rice cutgrass (*Leersia oryzoides*), and switchgrass (*Panicum virgatum*). Free-floating plants are common, including water hyacinth (*Eichhornia crassipes*) and common bladderwort (*Ultricularia macrorhiza*).

In the upper intertidal zone various species (Tiner 1993) of ferns, including royal fern (*Osmunda regalis*), sensitive fern (*Onoclea sensibilis*), and marsh fern (*Thelypteris thelypteroides*); sedges, including flatsedge (*Cyperus odoratus*) and threeway sedge (*Dulichium arundinaceum*); and shrubs, such as swamp rose (*Rosa palustris*), buttonbush (*Cephalanthus occidentalis*), and smooth alder (*Alnus serrulata*), may occur. A single marsh may support more than 50 species of wetland plants (Broome 1990).

Faunal Community

The diverse vegetation of tidal freshwater wetlands provides multiple niches for exploitation by fauna, especially benthic and epiphytic aquatic invertebrates. Larval and adult insects, amphipods, isopods, oligochaete worms, and mollusks are the dominant benthic macroinvertebrate taxa present. The surfaces of submerged and floating aquatic plants are colonized by a variety of epiphytic forms, especially aquatic insect larvae (Findlay et al. 1989). A diverse meiofaunal community exists, and the distribution of taxa is largely determined by microtopographic features (Yozzo and Smith 1995). Intertidal pools and rivulets are dominated by microcrustaceans such as ostracods and copepods. These shallow microhabitats are frequented by larval and juvenile fishes (i.e., *Fundulus* spp.) that forage intensively on the resident meiofauna. Distinct vegetative hummocks formed by marsh macrophytes such as arrow-arum support dense populations of nematodes, tardigrades, and naidid oligochates.

Decomposition of plant material occurs more rapidly in tidal freshwater wetlands than in salt marshes. Broad-leaved emergent perennials such as arrow-arum, *Pontedaria*, arrowheads (*Sagittaria* spp.), and *Nuphar* are characterized by high nitrogen concentrations, compared to *Spartina*, which has a higher carbon to nitrogen ratio (Odum and Heywood 1978). The resultant detritus is of greater nutritional quality to detritivores, such as amphipod and isopod crustaceans (Odum et al. 1984). Marsh plants also are consumed directly by higher taxa. For example, the seeds and stems of spike rush (*Eleocharis* spp.) are eaten by waterfowl and muskrats (Silberhorn 1976), and the seeds of smartweed are valued as a food source by ducks. Direct grazing of vascular plants is more prevalent in tidal freshwater wetlands than in salt marshes (Odum 1988).

Community composition of ichthyofauna is extremely diverse and is composed of predominantly freshwater species. This diversity is further enhanced by estuarine species that extend their range into freshwater habitats and anadromous marine species that use tidal freshwaters as spawning and nursery habitat. Resident freshwater species are represented primarily by three families: cyprinids (minnows and shiners), centrarchids (black bass, sunfishes, and crappies), and ictalurids (catfishes). Estuarine residents include white perch (*Morone americana*) and hogchokers (*Trinectes maculatus*). Migratory species occurring in tidal freshwater wetlands include juvenile striped bass (*Morone saxatilis*), menhaden (*Brevoortia tyrannus*), and summer flounder (*Paralichthys dentatus*) (Odum et al. 1979). The tidal freshwater wetland surface is used as a spawning habitat and nursery for several small marsh-resident species, including mummichogs and mosquitofish (*Gambusia affinis*). A diverse assemblage of freshwater and estuarine nekton species access the marsh surface to forage on flood tides (McIvor and Odum 1988). Lush beds of submerged vegetation in tidal creeks provide feeding habitat and a refuge from predators for numerous species (Rozas and

Odum 1987), including bluespotted sunfish (*Enneacanthus gloriosus*) and banded killifish (*Fundulus diaphanus*).

Reptile and amphibian communities are considerably more diverse in tidal freshwater wetlands relative to salt marshes, where most herpetofauna are excluded by saline conditions. Representative taxa in tidal freshwater include water snakes (*Nerodia* spp.) tree frogs (*Hyla* spp.), and a variety of turtles, including snapping turtles (*Chelydra serpentina*). The American alligator (*Alligator mississippiensis*) is a common inhabitant of southeastern and Gulf coast tidal freshwater wetlands and swamps.

Avian diversity is high in tidal freshwater wetlands, although waterfowl appear to use the marsh primarily as a food source, rather than for nesting (Odum 1988). Dabbling ducks and Canada geese (*Branta canadensis*) feed in tidal freshwater wetlands as they migrate south during late fall and early spring (Mitsch and Gosselink 1993). Great blue herons (*Ardea herodias*) are conspicuous year-round residents in Atlantic coast tidal freshwater wetlands. Soras (*Porzana carolina*) and other rails gather in large flocks in late fall to feed on wild rice (Odum et al. 1984). Tidal freshwater wetlands may also function as important temporary habitats for numerous arboreal bird species during migration periods.

Mammalian diversity (Stone et al. 1978; Odum et al. 1984) in tidal freshwater wetlands is high relative to salt marshes, although many of the same species (e.g., raccoons, rice rats) occur in both ecosystems. Muskrats (*Ondatra zibethicus*) and nutria (*Myocastor coypus*) are notorious for destroying large areas of marsh vegetation as a result of their foraging and nesting behavior. The latter is an introduced species that has spread throughout the Gulf coast and into the southeast in recent years (Mitsch and Gosselink 1993). Beaver (*Castor canadensis*) may be abundant in the upper reaches of many coastal rivers; however, the effect of their lodge-building activities on hydrology and community dynamics of tidal freshwater wetlands is largely unknown.

Mangrove Forests

Mangrove forests are made up of woody trees and shrubs, most notably within the family Rhizophoraceae, and are found in tropical and sub-tropical locales. As in salt marshes, tidal inundation is a major factor that structures the distribution of mangrove species. Unlike salt marsh ecosystems, however, mangrove forests are located only in areas of high temperatures (yearly average > 66°F) and humidity and are therefore not as widely distributed in the United States. Mangrove forests can occur within a wide range of salinities (optimally 10–20 ppt) but are less dense in low-salinity areas because of increased competition from freshwater plant species.

Geographic Distribution

Mangrove ecosystems in the United States are located primarily along the southwest Florida coast, although less extensive stands can be found in Louisiana, Texas, and Hawaii.

Zonation Within Habitats

The range of tidal flooding is largely responsible for the delineation of distinct zones of mangrove forests. Smooth cordgrass inhabits shallow intertidal areas (less commonly turtle grass [*Thalassia testudinum*] and manatee grass [*Syringodium filiform*] inhabit subtidal areas) seaward of mangroves. The red mangrove (*Rhizophora mangle*) is the most seaward species of woody vegetation primarily because of its tolerance of flooding. The black mangrove (*Avicennia nitida*) has a greater tolerance to exposure than the red mangrove and is found in the mid- to upper-intertidal zone. Succeeding the black mangrove is the white mangrove (*Laguncularia racemosa*), which grows primarily in the high-intertidal and above. These species are ultimately replaced by a variety of woody plants in supra-tidal and adjacent upland zones.

Northward from the southern tip of Florida and extending to Cedar Key, Florida (Kruczynski 1982), the mangrove ecosystem is made up primarily of black and white mangrove. Black mangrove is more salt tolerant and can survive moderate flooding, while white mangrove is most abundant in shallow, brackish water. Sparse red mangrove, often a pioneer species, occurs lower in the intertidal zone.

The only permanent stands of mangrove along the western Gulf of Mexico are found south of Corpus Christi (Webb 1982) and are made up of black mangrove, which is the most cold-tolerant of the North American mangrove species. Occasional stands develop north of Corpus Christi and in Louisiana, but periodic freezes kill off the plants.

Based on the periodicity of tides, riverine influence and terrestrial runoff, floral composition, and forest structure, six types of mangrove communities are commonly recognized (Lewis et al. 1985a): 1) basin forests, which are located along inland channels of terrestrial runoff; 2) hammock forests, which are similar to basin forests but occur at higher elevations; 3) riverine forests, which occur along river floodplains; 4) fringe forests, which are located along coastal areas and large islands where high elevations prevent regular tidal flushing; 5) overwash forests, which occur in areas of low elevation that are subject to regular tidal flushing; and 6) dwarf, or scrub, forests, which occur in areas of low elevation and nutrient-poor substrata. Red mangrove dominates in all these community types except basin forests, which are dominated by black mangrove (Lewis et al. 1985a).

Biological Community

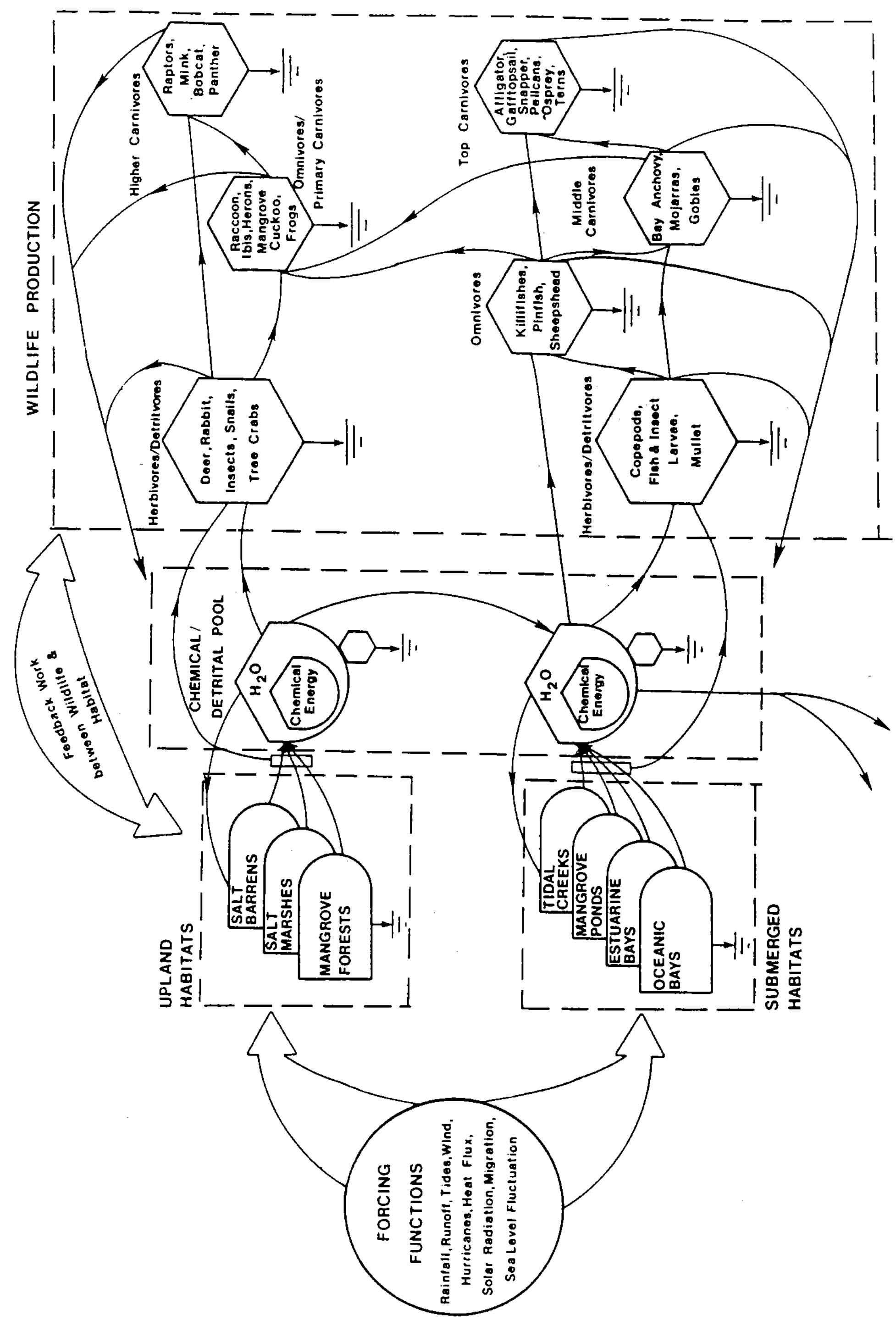
Mangrove trees form the basis of the food web in mangrove forests. Most primary production is manifested in the leaves of the plants. After senescence, leaves are colonized by bacteria and fungi and energy continues through the system in the form of detritus. Major detritivores here include copepods, amphipods, and the larvae of insects (Figure 5C-3). Populations of epiphytic algae are also primary producers of the ecosystem. These populations exist in mangrove forests by colonizing roots and the substratum and are grazed on by many animals, including the mangrove periwinkle (*Littorina angulifera*), which colonizes roots and branches above tidal levels. Nematodes and copepods represent the meiofaunal community.

Many species of fish use the root system for refuge and as a feeding ground (Odum and Heald 1972; Thayer et al. 1987), including red drum (*Sciaenops ocellata*), spotted seatrout (*Cynoscion nebulosus*), tarpon (*Megalops atlantia*), and snook (*Centropomus undecimalis*). Many species apparently leave the root habitat at night to feed in nearby seagrass beds (Thayer et al. 1987), although it appears that the abundance and overall diversity of ichthyofaunal communities within the mangrove ecosystem are generally higher than in seagrass beds. Fish community composition of the mangrove ecosystem is thus made up of both resident species (e.g., *Fundulus* spp.) and transients that move into and out of the root habitat (e.g., spotted seatrout).

The roots of mangroves are often covered by mangrove oysters (*Ostrea frons*). At higher elevations, fiddler crabs construct burrows in the spaces between roots. Common predators that inhabit mangrove stands include the crown conch (*Melongena corona*) and raccoons (McConnaughey and Zottoli 1983). Alligators and the endangered American crocodile (*Crocodylus acutus*) are also conspicuous predators here. Numerous bird populations are supported by these ecosystems (Lewis 1982) and overall avifaunal diversity appears to be greater here than in salt marshes.

Seagrass Beds

Seagrasses are those aquatic plants whose life functions, including flowering and pollination, occur underwater. These wetland plants are most tolerant to salinity



Source: Schomer and Drew (1982).

FIGURE 5C-3. Diagram of energy flow through the mangrove community.

and are the only truly marine angiosperms discussed herein. The major seagrass families of North America are Hydrocharitaceae, Zannichelliaceae, and Zosteraceae.

Geographic Distribution

Seagrass beds extend from New England to Florida along the Atlantic coast, along the eastern and western Gulf of Mexico, and from Washington to California along the Pacific coast.

Zonation Within Habitats

Seagrass beds of the Atlantic coast are made up primarily of eelgrass (*Zostera marina*), with widgeon grass (*Ruppia maritima*) occurring in shallow areas and in the intertidal zone. In southern latitudes, shoalgrass (*Halodule wrightii*) often replaces widgeon grass. Tropical and sub-tropical species occur south of the Carolinas, where beds are made up primarily of turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), shoalgrass, and widgeon grass (Thayer et al. 1979). Turtle grass is the dominant species. It may be mixed with shoalgrass or widgeon grass in shallow areas and with manatee grass subtidally. All of these species may be found mixed to a depth of about 10 m, below which manatee grass dominates to about 15 m (Zieman 1987). The most shade-tolerant of North American seagrasses, *Halophila* spp., may occur to depths greater than 50 m (though most common to about 30 m) in monotypic stands. It exhibits several adaptations for maximizing production under low levels of ambient light (Josselyn et al. 1986), including a high ratio of above- to belowground biomass, rapid turnover of leaf material, and rapid colonization of bare substratum under favorable light conditions. In shallow areas, these adaptations permit survival of *Halophila* spp. in turbid waters or during blooms of epiphytic algae that can be caused by nutrient runoff and other forms of perturbation, thereby serving as a potential indicator of environmental disturbance. This species is also found in mixed beds, along with turtle grass and manatee grass.

Seagrass beds in the eastern Gulf are made up predominantly of turtle grass, manatee grass, and shoalgrass, with shoalgrass mostly found in shallow areas. The majority of the beds are composed of turtle grass and manatee grass in various mixtures (Iverson and Bittaker 1986) to a depth of about 10–15 m. *Halophila decipens* and *Halophila engelmanni* are deep water, annual seagrasses and may be mixed with stands of turtle grass and manatee grass. In shallow areas near the mouths of rivers and in brackish water, widgeon grass often occurs in place of shoalgrass.

Seagrass beds of the western Gulf of Mexico include the species found in the eastern Gulf. Turtle grass, manatee grass, shoalgrass, and widgeon grass account for most of the plant biomass in this area. The size and extent of the beds are not as great as are found in the eastern Gulf, however. The zonation patterns of these species are generally the same as in the eastern Gulf and off eastern Florida.

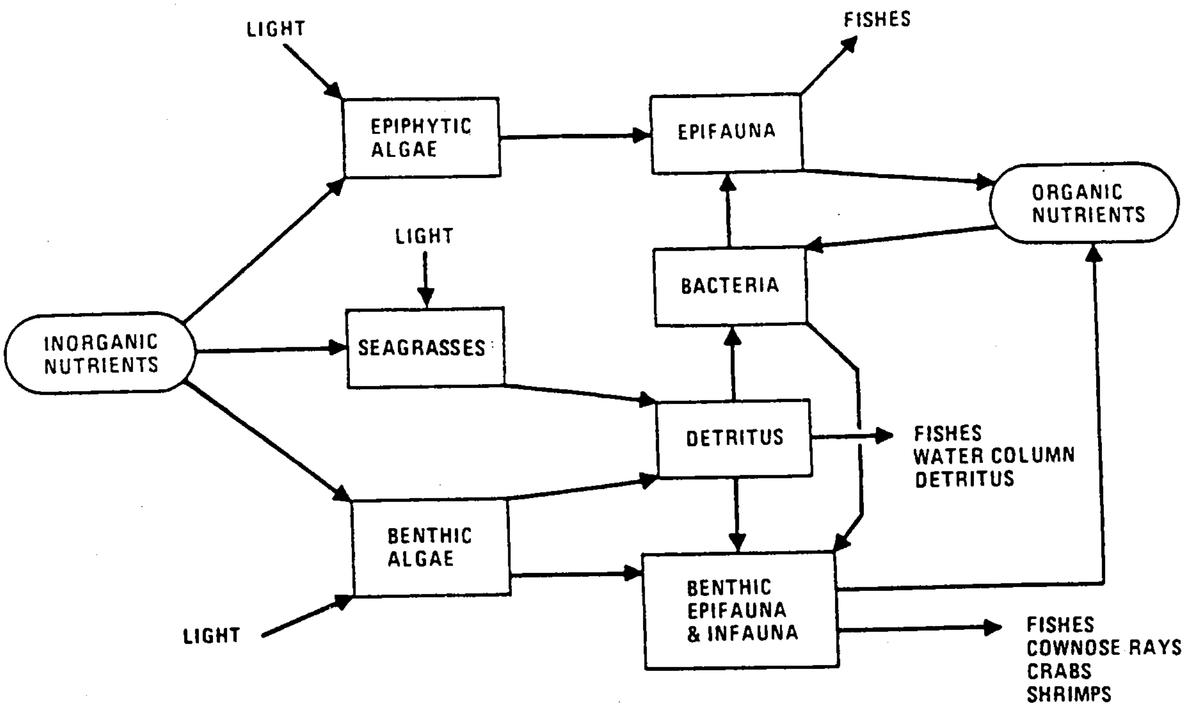
The dominant seagrass of California and the Pacific Northwest is eelgrass (*Z. marina*), with species of *Phyllospadix* occurring in the intertidal and shallow subtidal zones (Thayer et al. 1979).

Biological Community

Seagrass beds are found mostly below the intertidal zone. This is the primary reason for the higher animal diversities found in these habitats, relative to the exposed salt marshes and mangroves. Only a few animal species graze on live seagrass (Thayer et al. 1984); however, many of these species are increasingly threatened, including the green turtle (*Chelonia mydas*) and the Caribbean manatee (*Trichechus manatus*). As with other estuarine ecosystems, most plant production is consumed as detritus (Figure 5C-4). A more important energy source involves epiphytic algae, which colonizes the blades of seagrasses and is responsible for about 20 percent of the annual amount of primary production in seagrass ecosystems (Phillips 1982). These epiphytes include diatoms and both filamentous red algae and encrusting coralline algae. It has been suggested that seagrasses depend on the removal of fouling organisms by grazing epifauna, to prevent vegetative die-off resulting from shading (Van Montfrans et al. 1984). It is no surprise that destruction of established grassbeds may occur as a result of algal blooms induced by high levels of nutrient runoff.

The infaunal communities within seagrass ecosystems are usually dominated, in numerical terms, by tube-dwelling polychaetes (including Spionidae and Capitellidae), oligochaetes (Tubificidae), and pericarid crustaceans (Ampeliscidae and Idoteidae). Gastropods and bivalves, often including quahog (*Mercenaria mercenaria*), dominate infaunal biomass. The root-rhizome mat provides protection from predation for all infaunal species, with larger animals and deep-burrowing species protected to an even greater degree (Orth et al. 1984). This has been demonstrated in caging experiments where exclusion of predators from unvegetated areas resulted in population abundances and diversity similar to those found within the bed itself (Peterson 1982; Summerson and Peterson 1984).

Epifaunal assemblages are remarkably abundant and diverse when compared to adjacent unvegetated areas (Stoner 1980; Lewis and Stoner 1983; Lewis 1984). This is especially true for motile, macroinvertebrate ($\geq$1.0 mm total length)



Source: Green (1978).

FIGURE 5C-4. Conceptual model of seagrass communities.

species, especially within the Crustacea, which, as a group, exhibit distributions that are significantly and positively associated with vegetation density (Stoner 1980; Leber 1985; Heck et al. 1989; Williams et al. 1990; Thibaut 1992). The decapod crustaceans provide the major pathway between primary consumers and higher trophic levels (Virnstein et al. 1983). Ichthyofaunal communities also tend to be significantly associated with seagrass relative to unvegetated areas (Orth and Heck 1980; Heck et al. 1989).

As in other estuarine and coastal wetland ecosystems, many commercially important species depend on seagrass beds as a nursery ground for maturation of postlarval and juvenile individuals. Blue crabs are highly dependent on seagrass beds (Thomas et al. 1990; Williams et al. 1990) during vulnerable juvenile stages, as are brown shrimp. Many resident and transient marine fishes are dependent on seagrass beds, including recreationally important species such as spot (*Leiostomus xanthurus*), red drum (*Sciaenops ocellatus*), and spotted seatrout (*Cynoscion nebulosus*) (see *Open Coastline and Near Coastal Waters* for a detailed discussion of fish communities within temperate and sub-tropical seagrass beds).

The presence of submerged grassbeds provides a structural complexity not found in unvegetated substrata, yielding both food resources and refuge from predation. Seagrass ecosystems contain highly dynamic communities in which predator-prey and competitive relationships are mediated by vegetative structure. Faunal diversity is high, relative to periodically exposed intertidal habitats, primarily because of greater environmental stability in the permanently inundated subtidal zone.

KEY ECOLOGICAL PROCESSES

Nutrient Detrital Sources and Distribution

Estuarine and coastal wetlands are self-sustaining ecosystems that cycle nutrients vital to the health of plant populations. Vascular plants take up nitrogen, phosphorus, and other nutrients for life functions and growth. Both nitrogen and phosphorus are major components of river discharge from upstream runoff. The highest concentrations of dissolved and particulate nutrients imported into estuaries generally occur during periods of high freshwater flow, often as the result of springtime rains (Ward and Twilley 1986). The influx of nutrients also appears to be directly related to inundation time during the tidal cycle (Jordan et al. 1986). Generally, most of the nutrition needed for plant growth and function comes from particulate nitrate in the sediment and not from dissolved organic compounds in the water column. One reason for the high productivity of marsh plants is their ability to store excess nitrate for future growth. These nutrients are released back into the ecosystem via the decay of plant material and denitrification by bacteria. All wetlands trap and recycle nutrients.

Wetland plants also exhibit a tendency to absorb heavy metals and toxins (Kadlec and Kadlec 1979). Materials such as copper and zinc are taken up through the roots and incorporated into plant tissues. The degree to which these substances remain in the food chain may vary considerably, however. Thus, as water passes through estuarine and coastal wetlands, the concentration of these substances is reduced. These systems therefore act as natural water filters.

Detrital Processing and Nutrient Regeneration

Salt marsh ecosystems are areas of high primary productivity and may produce as much as 3 kg organic material/m^2-year (30,000 kg/ha-year) (Woodhouse and Knutson 1982). Although primary productivity is not as high within the mangrove ecosystem as salt marshes (Thayer et al. 1979), these woody forests may produce more than 1,000 g carbon/m^2-year (10,000 kg/ha-year), depending on the species of mangrove. Primary productivity of seagrasses is high, with tropical species such as turtle grass producing up to 4,000 g carbon/m^2-year (40,000 kg/ha-year) (Phillips 1982). In a review of literature on primary productivity in freshwater wetlands (Richardson 1979), tidal marsh vegetation produced biomass at the yearly rate of 16.2 kg/m^2-year (16,200 kg/ha-year).

In all of these ecosystems, detritus is the base of a food chain important to higher trophic levels. Detrital production releases nutrients into the food chain and is, therefore, important to processes of nutrient cycling in estuarine and coastal wetlands. The breakdown of plant material is initiated by microbial decomposition and herbivory.

Epiphytic algal populations are also important primary producers in all estuarine and coastal wetlands. Epiphytes provide the base of a distinct food chain that is used by wetland-inhabiting herbivores.

Despite the high productivity of these ecosystems, however, faunal diversity is generally low. This is because of fluctuations, both daily and seasonally, in temperature, tidal inundation, and salinity that few species can tolerate. These periodic fluctuations in the physical environment are disturbances that both define and restrict the composition of the plant and animal communities inhabiting coastal areas. Water salinity fluctuates diurnally and seasonally and is affected by tidal surge, river discharge, and surface evaporation from the marsh. Sediment characteristics are highly variable also, even within the same marsh. Most marsh plants are capable of surviving in a wide range of soil types, including sandy soils and clays. The duration of tidal inundation is also a limiting factor for marsh plants, and those located in the lower intertidal zone are well adapted to tidal fluctuation.

FUNCTIONAL VALUES

The presence of vegetation greatly aids in the control and prevention of coastal erosion, which is a major objective of marsh development projects. Emergent and submergent vegetation tends to slow currents and dissipate wave energy, thereby allowing for the deposition of suspended sediments. Accretion of sediments, rather than erosion, is thus enhanced. Vegetation further provides a dense root mat that tends to bind sediments and add stability to the shoreline. In certain natural situations, marsh establishment may actually be the result, rather than the cause, of low current energy and sediment accretion, and as sediments build up, colonizing plant species give way to other plants that thrive in areas at higher elevation.

ASSESSMENT OF HABITAT HEALTH

Degradation of estuarine and coastal wetlands ranges from subtle alterations, such as changes in vegetational standing crop or productivity, to more obvious and deleterious effects, such as large-scale filling of salt marshes. Key indicators of wetland degradation are discussed briefly below.

Physical Indicators

Physical indicators of habitat health include elevation, sediment texture, and turbidity.

Increased Elevation

Sedimentation and deposition of fill material result in increased elevation and altered hydrology (i.e., decreased tidal exchange and reduced saturation or flooding). Such habitat degradation is easily recognized in comparison with pre-alteration or surrounding conditions based on topography, development, or vegetation changes.

Decreased Elevation

Excavation in estuarine and coastal wetlands causes increased water depth, decreased circulation, and altered sediment texture. Degradation can be recognized by the presence of stagnant depressions, by the absence (removal) of vegetation, or by changes in vegetation type. For example, removal of sediment in black needlerush marshes creates habitat for saltgrass and bulrush (*Scirpus olneyi*). Excessive deepening creates unvegetated habitats characterized by low dissolved oxygen and high organic accumulation.

Sediment Texture Changes

Wetland sediment composition may become coarser because of sedimentation caused by erosion, stormwater runoff, or storm-induced wave action. This type of degradation will be visible in surface deposits of sandy material. Finer sediments may also occur through stormwater runoff and are usually is related to construction or development activities such as roadbuilding, dredging and dredged material disposal, and land clearing and earthmoving.

Increased Turbidity

Increased suspended sediment load related to coastal development may cause wetland degradation, especially in seagrass ecosystems. Increases in turbidity caused by dredging activity has been observed to destroy turtle grass beds (Phillips 1980) both by decreasing light penetration and physically smothering the vegetation. Degradation may be observed as the persistent presence of cloudy water.

Biological Indicators

Biological indicators of habitat health include the presence of barren zones, excessive epiphytic growth, invasion, of non-marsh species, proliferation of weedy species, and absence of key fauna.

Presence of Barren Zones

Many estuarine and coastal wetlands show the effects of habitat degradation by the presence of bare zones where vegetation had previously been established. These unvegetated areas may be a result of either biological or anthropogenic perturbations.

In seagrass ecosystems, seagrass has been found to be smothered by the excavations of burrowing shrimp (*Callianassa* spp.) (Suchanek 1983). It has been documented that feeding activity by cownose rays can physically damage seagrass beds (Orth 1977). In salt marsh ecosystems, the presence of barren areas may be caused by several natural factors, including natural hypersaline conditions, erratic tidal fluctuation (Brown et al. 1976), and storm-induced deposition of *Spartina* wrack (Reidenbaugh and Banta 1980).

Unnatural disturbances account for most of the loss of estuarine and coastal wetland habitat. Nutrient loading caused by excessive runoff can cause rapid epiphytic and phytoplankton growth, thereby shading seagrasses and causing death due to a lack of photosynthetic activity (Bittaker 1975). Seagrass meadows may also sustain damage by outboard engine propellers (Zieman 1976), with tracks persisting up to several years after initial damage. Other perturbations can include the presence of leachate from industrial activities, the development of levees and channels, and the presence of toxic materials such as oil and other spilled chemicals. It has been estimated that more than half of North America's original salt marshes and mangrove forests have been destroyed (Watzin and Gosselink 1992), mostly by filling for land creation. In California alone, an estimated 90 percent of original wetland habitat has been destroyed (Watzin and Gosselink 1992).

Excessive Epiphytic Growth

Epiphytes (plants that are attached to other plants) are most prevalent in areas of stressed seagrass populations. The presence of excessive epiphytic growth on seagrasses usually indicates habitat alteration or degradation from nutrient enrichment and/or increased suspended solids and decreased light penetration. Bittaker (1975) looked at offshore seagrass productivity in Apalachee Bay, Florida, in relation to the polluted Fenholloway River estuary and found decreased seagrass productivity as a result of high turbidity and increased phytoplankton productivity caused by high nutrient levels.

Invasion of Non-Marsh Species

Increased abundance of non-marsh plant species reflects habitat degradation. Thus, the incursion of slash pine (*Pinus elliottii*) in brackish marshes or the expansion of saltmeadow cordgrass into needlerush zones generally indicates areas of sedimentation or drainage.

Proliferation of Weedy Species

Increased predominance of weedy species such as common reed (*Phragmites australis*) and wax myrtle (*Myrica cerifera*) indicates habitat changes related either to clearing, filling, or draining activities in tidal marshes (Rejmankova 1992). Such species will crowd out other resident species and decrease habitat wildlife values and diversity.

Absence of Key Fauna

Reductions in the overall diversity of estuarine and coastal wetland ecosystems may indicate habitat degradation. The disappearance of common resident species, such as fiddler crabs (*Uca* spp.), may be symptomatic of further wetland degradation. Shellfish populations are very susceptible to degraded water quality and increased turbidity, and declines in these populations also may be indicative of a reduction in habitat quality. The depletion and loss of commercial fisheries in many parts of the United States are attributed to the destruction of estuarine and coastal wetland environments.

KEY ENVIRONMENTAL PARAMETERS

Primary Factors

Primary environmental factors in estuarine and coastal wetland distribution, survival, and growth include the physical parameters of water depth, circulation and currents, turbidity, and substrate quality and the chemical parameters of salinity and redox potential. Each of these parameters is described in Table 5C-1 and below.

Water Depth

Water depth and duration of tidal inundation together are a major determinant of zonation, particularly in smooth cordgrass marshes on both the Atlantic and Gulf of Mexico coasts. Depth is also a critical determinant of the species composition and distribution of seagrasses: decreased light intensity with water depth is a limiting factor for seagrasses. Species such as *Halophila* spp. are adapted to deeper waters, while widgeon grass is found in shallower water. In mangrove ecosystems, the flood-tolerant red mangrove occurs in low- to mid-intertidal zones. As elevation increases, red mangroves are replaced by black and then white mangrove plants.

TABLE 5C-1. KEY ENVIRONMENTAL PARAMETERS IN ESTUARINE AND COASTAL WETLANDS

Parameter	Comment
Physical	
Water depth	Water depth (including temporal patterns of tidal inundation) is a major determinant of zonation in coastal wetlands (e.g., subsidence results in deeper waters in wetlands and increased elevation can reduce frequency and depth of tidal inundation).
Circulation and currents	Currents and tidal flushing control nutrient flux and waste removal. Circulation patterns, current velocity, and habitat dynamics are often altered if anthropogenic structures are introduced into the environment.
Turbidity	Turbidity is a key factor in the distribution of seagrasses because of its effect on decreased light energy for photosynthesis. Estuarine waters are more turbid than marine waters.
Substrate quality	Sediment composition (e.g., grain size, organic content) affects the ability of marsh plants and seagrasses to establish roots and obtain nutrients and dissolved oxygen. Coarse or unstable sediments do not allow for rooting of wetland vegetation. Fine sediments typically contain high organic content and low dissolved oxygen.
Chemical	
Salinity	Interstitial and water column salinities are major determinants of marsh and seagrass community composition. Distributions of most coastal wetland species are related to a salinity regime.
Redox potential	Redox potential is related to organic content of wetland sediments and extent of oxygenation, water quality effects on pH, nutrient enrichment, and plant inhibitors.
Biological	
Competition between plant species	Competition between wetland plant species can be an important factor in their distribution and abundance.
Plant and animal interactions	Plant-animal interactions (e.g., herbivory) may also limit distributions of coastal wetland plant species.

Circulation and Currents

The presence of currents and tidal flushing results in nutrient flux and waste removal. Species such as smooth cordgrass are successful mainly in areas exposed to strong tidal flushing and moderate currents: East coast marshes, where tidal fluctuations are wide, contain broader smooth cordgrass zones than Gulf of Mexico salt marshes, where flushing is limited to margins of tidal creeks and channels created by small tidal amplitude. Mangroves are found primarily in areas of good tidal flushing. Seagrasses are also affected by circulation: areas of greater tidal flushing are less likely to experience sedimentation of silts and clays. Species that are intolerant of siltation and smothering, such as turtle grass, are found primarily in such areas.

Turbidity

Turbidity, or conversely water clarity, is a key factor in the distribution of seagrasses. The availability of light energy for photosynthesis decreases with increased turbidity and restricts the depth of seagrass growth in coastal waters. Estuarine waters are more turbid than marine waters; therefore, seagrasses are restricted to a depth of less than 2 m in most estuaries but may occur at depths over 85 m in clear oceanic waters (den Hartog 1970).

Substrate Quality

Sediment texture and composition (grain size, organic content) affects the ability of marsh plants and seagrasses to establish roots and obtain nutrients and dissolved oxygen. Fine sediments typically contain high organic content and occur in areas of limited tidal exchange. These habitats are vegetated by species such as black needlerush, which are tolerant of low dissolved oxygen and limited tidal flushing. On the other hand, very coarse and unstable sediments do not provide suitable habitat for estuarine and coastal wetland vegetation.

Salinity

Interstitial and water column salinities are major determinants of marsh and seagrass community composition. As described earlier, distributions of most estuarine and coastal wetland species are at least partially related to salinity regime. Brackish water habitats support marsh species such as black needlerush and saltgrass, black mangroves, and seagrasses such as manatee grass and shoalgrass. Saline conditions favor marsh species such as Pacific cordgrass and smooth cordgrass, mangroves such as the red mangrove, and seagrasses such as turtle grass and *Halophila* spp. Saline water limits plant competition with halophytes such as smooth cordgrass and mangroves, resulting in monotypic stands of these species. Freshwater habitats lack the stress of salinity and contain a wide variety of species, including wild rice, bulrush, spike rush, and sedges.

Redox Potential

The extent to which wetland sediments are reduced or oxidized is affected by circulation/flushing and sediment organic content. Species such as black needlerush and black mangrove occur in low-energy waters where sediments are relatively fine and organic-rich. Roots of such species are tolerant of low interstitial dissolved oxygen as well as elevated sulfides. Species that are less tolerant of these conditions typically occur in areas of high tidal exchange, strong currents, and sandier sediments. The degree of soil reduction and elevated levels of sulfides in salt marsh sediments has been found to be positively correlated with variation in the growth form (e.g., tall or short) of smooth cordgrass (Mendelssohn et al. 1981).

Secondary Factors

Secondary environmental factors include habitat parameters such as sediment stability, tidal periodicity, water quality, and biological interactions. These parameters reflect conditions that are influenced mainly by the factors described above. For example, sediment stability/erosion potential is related to current velocity and tidal exchange in addition to substrate quality. Wetland species associated with high energy coastal habitats (e.g., red mangrove and turtle grass) are also associated with less stable (sandier) sediments. Estuarine habitats characterized by reduced currents and flushing, such as blackgrass and salt-meadow cordgrass, are adapted to infrequent tidal inundation and stable, fine-textured sediments. Similarly, wetland species that are sensitive to reduced dissolved oxygen and increased pH (e.g., smooth cordgrass, red mangrove, turtle grass) are associated with well-flushed estuarine or marine habitats.

Competition between wetland plant species can be an important factor in their distribution and abundance. For example, common reed, a very invasive species, spreads into new habitats relatively slowly. However, it can completely displace other marsh species in marginal habitats, such as disturbed estuary embankments and beach berms.

Plant-animal interactions may also limit estuarine and coastal wetland species' distributions. Sea urchins (*Arbacia punctulatus*) appear to affect the distribution of turtle grass through grazing. Nutria may open large expanses of brackish and freshwater marshes through grazing, creating habitat for a wider variety of wetland plants (Linsey and Linsey 1972). Muskrats are also important in opening marsh habitats to other species: saltgrass and other estuarine marsh grasses are able to spread into areas where muskrats have removed black needlerush.

RESTORATION PROJECTS

Potential restoration projects in estuarine and marine wetlands are summarized in Table 5C-2. The first column lists problems that may cause habitat loss, damage, or deterioration in each type of habitat. For each problem, possible solutions are presented in the second column.

Typical estuarine and marine wetland habitat alterations include the following:

- Hydrologic change caused by interruption of sediment input and ongoing subsidence (e.g., Mississippi River Delta)
- Hydrologic change from navigation channel construction and dredged material disposal (e.g., Galveston Bay and coastal Louisiana)
- Wetland loss caused by chemical contaminants (e.g., seagrass habitat degradation by pulp mill effluents)
- Wetland loss caused by alteration in turbidity, sedimentation, and sediment texture.

These types of alterations represent the majority of saltwater wetland habitat problems that may be the focus of future restoration projects. Four different principal project types related to the above alterations are presented below.

Project Type 1: Hydrologic Regime Restoration

In areas where normal hydrology has been changed through diversion of flow patterns, restoration can involve removal of constructed berms or opening of flow points through levees to allow renewal of sediment deposition from riverine floodwaters. It is expected that this restorative process would be gradual and that reestablishment of sedimentation would be accompanied by a change in salinity (to a less-saline condition) during periods of overflow. Gradual successional changes in the affected saltwater marshes would occur because of increased ground elevation and decreased salinity. An example of this type of restoration is the marsh restoration project near Grand Isle, along the west levee of the Mississippi River in Louisiana.

TABLE 5C-2. RESTORATION PROJECTS IN ESTUARINE AND COASTAL WETLANDS

Problem	Possible Solutions
Disturbed sedimentation regime due to altered hydrologic patterns	Remove constructed berms or open flow points through levees to allow renewal of sediment deposition from riverine floodwaters. The reestablishment of sedimentation would be accompanied by a change in salinity (to a less-saline condition) during periods of overflow, resulting in gradual successional changes in the affected saltwater marshes. Upstream sediment catch basins can be used to trap excess sediment delivered to coastal marshes.
Disruption of hydrologic patterns (e.g., caused by navigation channels)	Restoration requires a combination of filling channels, removing constructed berms, and replanting with native saltwater wetland species.
Fill material added to ecosystems	Remove fill to reestablish natural grades and normal hydrologic regime with or without appropriate vegetation or topsoil replacement.
Degradation of water and/or sediment quality	Eliminate or attenuate the contaminant input (e.g., through improved waste minimization technology). Remove contaminant sinks such as deposits of chemically degraded sediments. Relocate discharge location.

Another example of hydrologic restoration involves removal of dikes built to separate tidal marshes from the Indian River Lagoon estuary in southeast Florida. These mosquito control impoundments would be reopened to the lagoon in an attempt to restore saltmarsh habitat characteristics. In Connecticut (Sinicrope et al. 1990), the removal of previously constructed dikes resulted in a natural expansion of salt marsh vegetation, concurrent with a decline of non-invasive brackish water vegetation.

Project Type 2: Hydrologic Change Through Elimination of Navigation Channels

There are numerous estuarine and coastal wetland areas that have been damaged by excavation of canals, waterways, and similar navigation channels. In such areas, restoration involves a combination of channel filling, constructed berm removal, and replanting with native saltwater wetland species.

Such projects could be performed in locations like the Florida Keys, where mangrove wetlands have been excavated to create access channels and canals. If a suitable, stable substrate is recreated, mangroves can be replanted successfully and ecosystem functions, such as wave projection, erosion control, and faunal habitat, can be reestablished.

Other restoration efforts of this type could involve refilling oil field access canals in the Mississippi Delta area. Removing berms to refill such canals can be effective in reestablishing suitable saltmarsh habitat. Replanting these areas with species such as smooth cordgrass can speed the rate of marsh habitat restoration.

Project Type 3: Removal of Fill Material

Areas that have been filled for coastal development or for dredged material disposal could be restored through removal of the fill to reestablish natural grades and a normal hydrologic regime. In most cases, however, the old fill will have caused significant compaction/subsidence of the native marsh soil, and a new layer of suitable organic topsoil will need to be introduced to reestablish the appropriate marsh elevation and surface soil conditions. Generally, marsh restoration includes revegetation with appropriate species to reduce the time needed to reestablish marsh plant densities and ecological functions.

Project Type 4: Restoration of Water and/or Sediment Quality

Chemical degradation of saltwater wetland habitats could be restored through either elimination/attenuation of the contaminant input (e.g., through improved waste minimization technology), or through removal of contaminant sinks such as deposits of chemically degraded sediments. In the former case, improved water quality could be achieved through relocation of discharges, such as in certain seagrass areas of coastal Florida where paper mill effluents have caused reductions in turtle grass populations. Relocation of the discharge point and improvements in effluent quality, have been found to allow reestablishment of normal seagrass populations.

Removal of contaminated sediment could also result in restoration of saltwater wetlands. Extensive deposits of aluminum mill wastes in Polecat Bay, Alabama, have caused degradation of brackish marshes and beds of widgeon grass. Restoration of these wetlands requires careful removal of accumulated sediments to prevent dispersion of contaminants by resuspension, oxidation and other means. Special equipment would be needed to excavate deposits that could be as little as 0.5-ft thick, and excavated material would require containment in secure upland disposal areas. Replanting restored habitats may be necessary, depending on the areal extent of the restored site and distance form native plant populations.

Five case studies are presented below to illustrate restoration projects in estuarine and coastal wetlands. The first case study describes the restoration of island shoreline along Terrebonne Parish in central Louisiana. This area has been severely impacted by the combined effects of subsidence, erosion, and rise in sea level. The second example is the Florida Keys Seagrass Restoration Project, which was a phased project to reestablish seagrass meadows destroyed during road construction activities. The Los Peñasquitos Lagoon enhancement plan and program was developed to reduce habitat alteration or loss in this lagoon ecosystem in San Diego, California. Details of this project are presented in the third case study. The next case study describes a habitat mitigation and improvement project in Tampa Bay, Florida. This area has been adversely affected by the continuing development within its drainage basin. The final case study in this section examines the restoration of marshes impounded to create pasturelands.

Barrier Island and Back Barrier Marsh Reconstruction

Terrebonne Parish, located on the central Louisiana coast, has experienced severe land loss as the result of subsidence, erosion, and rise in sea level (Edmondson and Jones 1985). Coastal erosion rates of 15 m/year have been documented for the Isle Dernieres barrier system (Figure 5C-5) (Penland and Boyd 1981). Loss

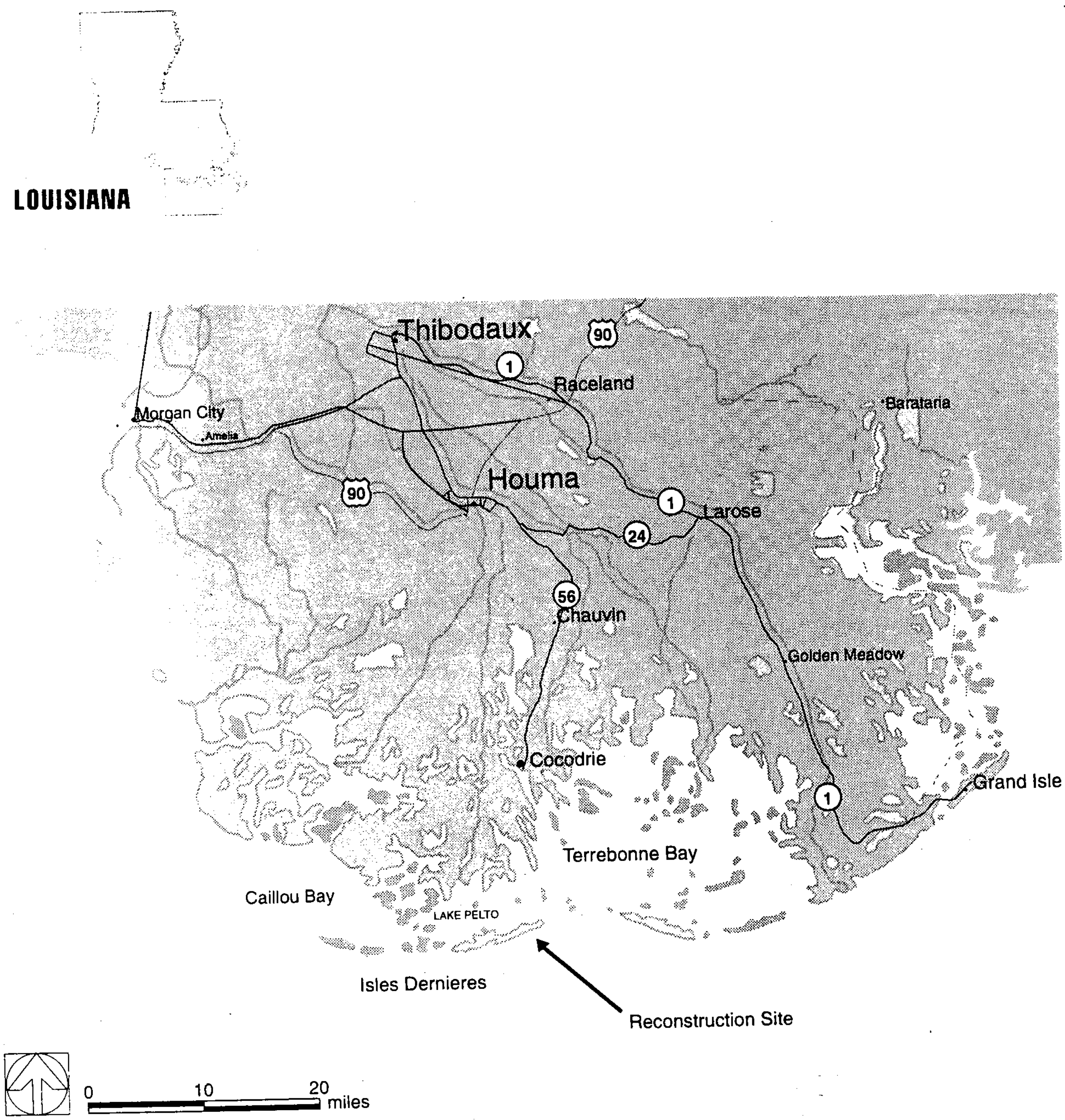


FIGURE 5C-5. Barrier island and back barrier marsh reconstruction site, Isle Dernieres (Terrebonne Parish).

of this island chain has serious implications for mainland land loss and for tidal freshwater marshes behind the barrier island.

The Terrebonne Parish project involved extensive planning, including public information and comment meetings, public agency participation, and scientific/ engineering community participation. The resulting plan comprised several restoration measures, including saltwater control dikes and closure system, canal bank reconstruction, beach nourishment to counter rapid shoreline erosion, dune construction, and sand retention measures. The principal objective of this plan was to reinforce the island shoreline via beach/dune reconstruction. A secondary objective was to reestablish hydrologic and salinity conditions essential to the success of tidal freshwater wetlands that comprise the mainland shoreline.

Restoration Approach

Erosion of the shoreline was to be reduced by nourishing the beach with coarse sediments, closing storm-created breaches and washovers, and installing sand-trapping jetties. Beach nourishment material was obtained from the U.S. Army Corps of Engineers' (Corps) Cat Island Pass dredging project. Approximately 300,000 yards of silty sand was pumped onto a 3,200-ft stretch of island beach-front to elevate a total of 38 acres of the barrier island.

Shoreline restoration efforts were completed in March 1985. By these efforts, a washover and potential breach were sealed. Use of overwash sand to rebuild the island introduced numerous seeds and cuttings, and vegetation quickly germinated on the restored dunes. Sand fences were then installed on the front dike/dune, and dune vegetation was planted on several acres of the restored area. A major storm passed through the area during dike construction, but it caused only minor damage to the containment system. The beach front quickly eroded to the angle of repose associated with the adjoining natural beaches. Closure of the island breach decreased salinities in ponds previously salinized by overwash, and waterfowl abundance increased in these areas. This effort was considered a success, although long-term management and monitoring were not pursued.

Evaluation of Restoration Efforts

This project was designed to correct damage to marshes by breaching of the barrier island, but it represented an ecosystem-level approach to planning. Marsh benefits were allowed to accrue indirectly through the success of island reconstruction, while the success of breach and washover closure included revegetation of the barrier island as well.

The contract scoping process included preliminary analysis of sediments in potential borrow areas and of beach and dune slopes at the reconstruction site. Nonetheless, potential bidders hesitated to submit fixed-cost bids for this work because of possible risks associated with long pumping distances and exposure to open Gulf of Mexico waters. An alternative borrow site was identified through discussions with various dredging contractors and the Corps.

After the contract was awarded, new information led to changes in project design, which were facilitated by flexibility in contract management. This flexibility contributed to the eventual success of this project. However, long-term monitoring would have improved the Parish's ability to measure ecological outcomes and success and to adjust restorative measures to changing conditions. This initial project has been followed by several on-going restoration projects in the barrier island chain, including Raccoon Island, Timbalier Island, Grand Isle, and Isle Dernieres.

For more information on this project, contact:

Terrebonne Parish Service Center
P.O. Box 2768
Houma, Louisiana 70361

Florida Keys Seagrass Restoration Project

Located between Florida Straits and Florida Bay, the Florida Keys experienced extensive loss of seagrass meadows during the replacement of 37 bridges between upper Matecumbe Bay and Key West (Figure 5C-6). Approximately 27 ha of turtle grass (*Thalassia testudinum*) were destroyed; however, of this 14.3 ha were considered restorable (Lewis 1987). Areas considered not restorable were deemed as such because of shading by bridges and loss of sediment structure after bridge construction.

The principal objective of the Florida Keys Seagrass Restoration Project was to re-establish seagrass meadows using shoalgrass (*Halodule wrightii*) long-shoots. Shoalgrass, rather than turtle grass, is more suitable for restoration projects because it has a greater potential for establishing grass beds. Natural succession may ultimately result in development of a turtlegrass commmunity.

Restoration Approach

This restoration project was executed in two phases. Phase I occurred during April 1983, with the planting of shoalgrass over a 9.1-ha area. Phase II occurred

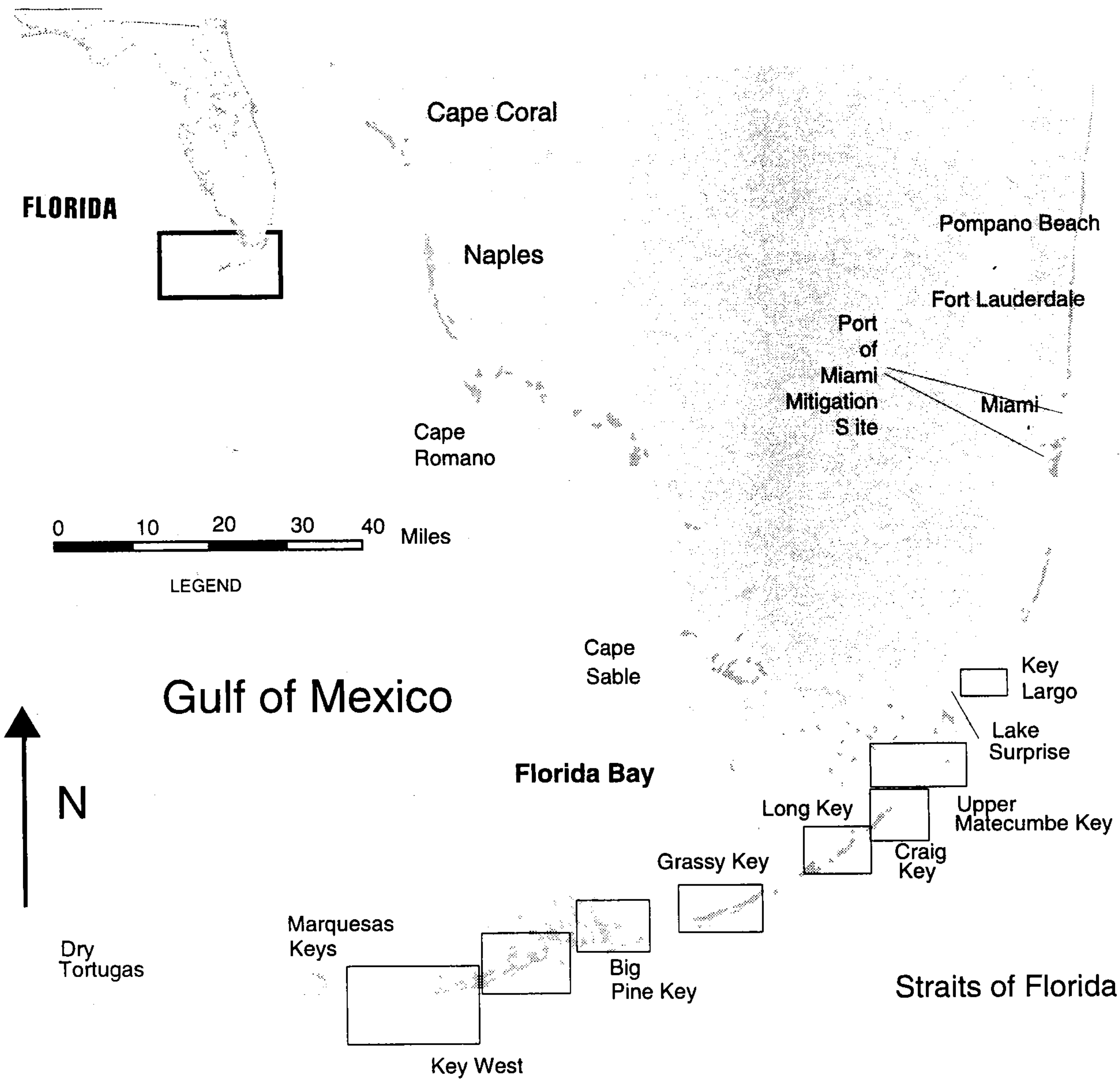


FIGURE 5C-6. Florida Keys seagrass restoration project.

in August 1983, with an additional 4.4 ha planted with either shoalgrass (most sites) or manatee grass (*Syringodium filiforme*). Shoots were planted on 1-m centers and anchored to the substratum with either staples or nails.

By August 1984, approximately 8.3 ha of shoalgrass had been successfully restored, equalling 61.3 percent of the original planted area. Compensatory planting subsequent to Phases I and II resulted in an additional 6.7 ha of shoalgrass establishment. The total surviving area of seagrass planted during all phases of the Florida Keys Seagrass Restoration Project equaled 14.0 ha, for a success of 72.8 percent (Lewis 1987). By August 1985, natural recruitment of turtle grass seedlings into shoalgrass beds was occurring.

Evaluation of Restoration Efforts

This project was designed to re-establish seagrass meadows destroyed by bridge construction using an initial planting of an easily established "colonizing" species (shoalgrass). Establishment of the original species (turtle grass) was then to occur naturally.

A long-term monitoring program at the project sites was established to monitor seagrass cover and persistence (Fonseca 1990). These criteria allowed verification of success over time that was beneficial to both the contractor and the regulatory agencies. This comprehensive monitoring allowed the contractor to plan for alternative methods of planting and to undertake compensatory planting if needed. Regulatory agencies were provided with a verifiable means of determining compliance.

For more information on this project, contact:

Lewis Environmental Services, Inc.
P.O. Box 20005
Tampa, Florida 33622

Los Peñasquitos Lagoon Enhancement Plan and Program

The Los Peñasquitos Lagoon, located north San Diego County, California (Figure 5C–7), presently comprises 636 acres of marsh, open water, and peripheral wetland habitats at the terminus of a 98 mile^2 watershed. The lagoon has been altered from a tidal estuary to a frequently isolated pooled system characterized primarily by salt marsh and shrubby to forested wetland (Los Peñasquitos Lagoon Foundation and State Coastal Conservancy 1985). The lagoon has suffered from two major problems: sedimentation and lack of tidal

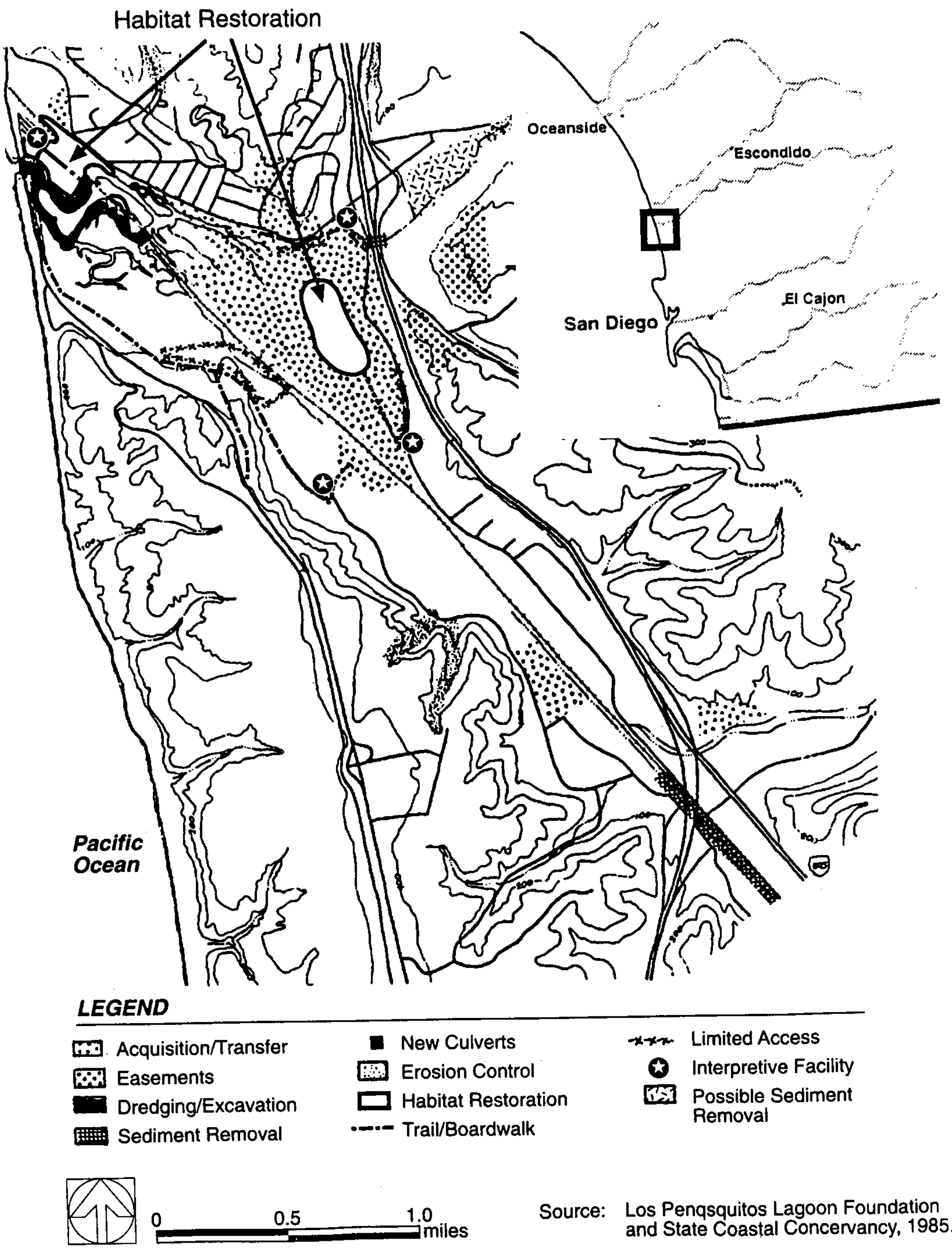


FIGURE 5C-7. Los Peñasquitos lagoon enhancement plan and program.

flow (Marcus 1989). Both processes have resulted from rapid and ongoing urbanization of this coastal watershed. Approximately 70,000 yd^3 of sediment are deposited in the lagoon each year. The elevation of the northeast corner of the lagoon increased by 6.1 ft from 1968 to 1985 (Los Peñasquitos Lagoon Foundation and State Coastal Conservancy 1985). These alterations have eliminated much of the habitat for several endangered species, resulted in severe changes in water quality and hydrography, and caused periodic fish kills.

The lagoon enhancement plan was developed by the State Coastal Conservancy and the Los Peñasquitos Lagoon Foundation, with assistance from the City of San Diego. Background information on physical processes and ecological structure of the lagoon was developed by the California Coastal Commission (Prestegaard 1978) and San Diego Association of Governments (SANDAG 1982) and focused on the need for watershed management to reduce sediment input to the lagoon and resulting habitat alteration or loss.

The general objective of the enhancement plan is "to protect, maintain, and enhance the Los Peñasquitos Lagoon system and adjacent uplands," to perpetuate the native biota associated with the lagoon, and to restore and maintain estuarine conditions that existed before anthropogenic activities altered the system (Los Peñasquitos Lagoon Foundation and State Coastal Conservancy 1985).

Restoration Approach

Lagoon enhancement measures have primarily involved periodic opening of the lagoon mouth to reestablish tidal flushing following closure by sediment accumulation. Monitoring of the effects of lagoon opening and closing on water quality and biota was also a major component of the project. Results of quarterly monitoring have been reported since 1987 and provide a basis for modifying restoration methods to fine-tune the plan. Other plan components include expanding park and open-space areas, improving circulation within the lagoon, restoring specific habitats in the system, providing public access, controlling sedimentation from watershed construction; and mitigating unavoidable wetland losses by developments. Secondary elements of the plan include cleaning up illegal dumps; strengthening existing grading ordinances; maintaining stream channels and basins in the watershed; acquiring key watershed properties and easements; placing culverts at strategic points; and constructing trails, boardwalks, and interpretive facilities for public education.

Evaluation of Restoration Efforts

This ecosystem-level project has been partially successful in restoring tidal flow to the lagoon system and in promulgating more effective controls on sediment transport from grading and other development activities in the watershed. The enhancement project has also been effective in securing properties and easements important to management of the lagoon. It has provided a good model for planning and establishing a community-based restoration plan that reflects long-term commitment and adaptive management opportunities. Each year, results of monitoring are used to identify variations in the plan approach, with recommendations for evaluation of issues not previously addressed. For example, Williams and Gibson (1995) suggested that cobbles be removed from under the bridge at the mouth of the lagoon during March and early April, at a minimum.

Monitoring results suggest that many biological communities within the lagoon system have not improved significantly since the project's beginning in 1987. Because the hydrologic characteristics of the system have been altered so severely, complete restoration may not be achievable. Sedimentation from continued development in the watershed is still a force in degrading the lagoon ecosystem, while the tidal prism under the highway is inadequate to keep the entrance to the lagoon open under most circumstances. According to Williams and Gibson (1995), exotic weeds are still a problem in southern lagoon areas. From these observations, the lagoon system does not appear to have reached an equilibrium, and the success of the plan cannot be fully assessed.

For more information on this project, contact:

Los Peñasquitos Lagoon Foundation
P.O. Box 866
Cardiff, California 92007

Tampa Bay Habitat Mitigation Improvement Project

Tampa Bay, Florida, has been adversely affected by the continuing development within its drainage basin, which includes filling and/or excavating portions of the bay bottom; alteration of area hydrology by bridges, roadfill, and runoff collection systems; and water quality degradation from nutrient enrichment, chemical contaminants, and sediment load. These problems were characterized in a symposium on the status of the Tampa Bay system held in May 1982 (Treat et al. 1985). That conference (known as "Tampa BASIS") addressed the water quality, emergent wetlands, fishery resources, and other key aspects of the estuarine ecosystems in the bay. Seagrass habitat had declined from 76,496 acres prior to 1876 to only 14,203 acres in 1982 (Lewis et al. 1985b). The shoreline and shallow bay fringe has been modified by residential development (as at Boca Ciega Bay) and by various port and highway projects. More than half of an estimated historic cover of 25,000 acres of marsh and mangrove was reported to have been lost by 1982 (Estevez and Mosura 1985). From 1963 to 1978, finfish landings decreased by roughly 30 percent (Lombardo and Lewis 1985).

Restoration Approach

Tampa Bay habitat restoration projects have been performed recently under the sponsorship of the Tampa Bay Regional Planning Council. The area of these restoration efforts is shown in Figure 5C-8.

One project initially involved transplanting seagrass in upper Tampa Bay, from a causeway expansion site to Rocky Creek (Clark 1989). Subsequent use of several other relocation sites was evaluated on the basis of key habitat parameters such as water depth, substrate type, and proximity to natural beds. Shoal grass and *Ruppia maritima* were planted using various techniques in each of six restoration sites. Seagrass collector stakes were used with mixed results: from 2 to 68 percent of the stakes captured drift seagrass sprigs within a 129-day period. Lack of success was attributed to epiphytic growth, poor water quality, and unsuitable sediment type.

Tidal marsh restoration in the Bay in 1991 involved planting *Spartina alterniflora* and *Paspalum distichum* in an artificially created tidal lagoon, using volunteer fishermen labor (Clark 1992). Marsh restoration was very successful, in terms of public participation as well as plant survival and growth. Seventeen community organizations were involved in design and implementation of this restoration effort. The tidal marsh achieved dense coverage 1 year after planting was completed, and a variety of fauna was observed in the lagoon and marsh.

Evaluation of Restoration Efforts

Seagrass restoration efforts were monitored annually; results of the preliminary transplant operation were used effectively to improve the likelihood of success 4 years later. The scope of this project was not an ecosystem-based approach because it focused on only two seagrass species. These projects minimally used an adaptive strategy, although lessons learned at these six sites could be applied to subsequent restoration efforts.

Similar management limitations applied to the marsh restoration project. However, construction of the tidal lagoon, coupled with marsh planting, produced ecosystem-level habitat restoration. Neither restoration project considered a safe-fail outcome scenario.

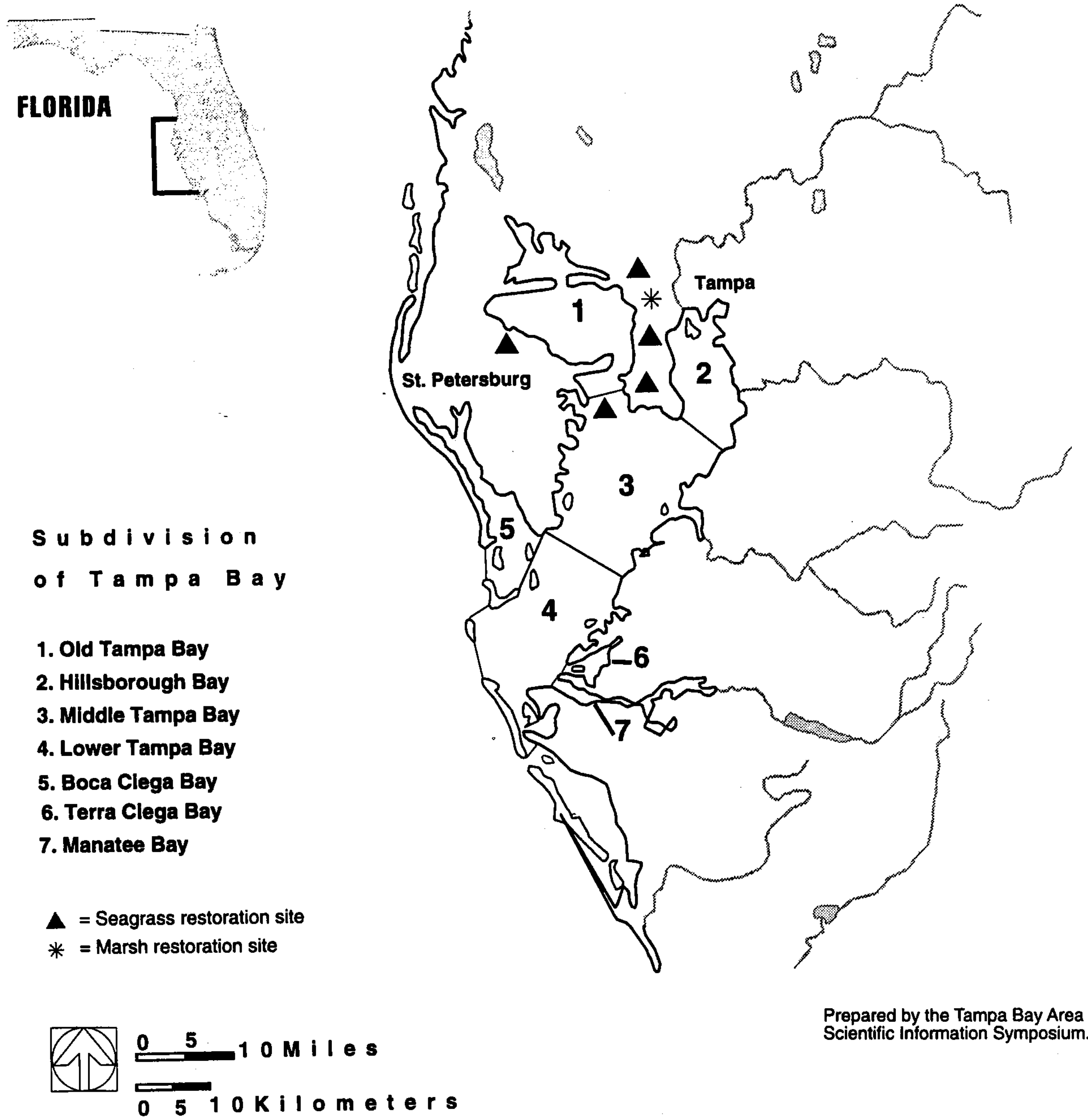


FIGURE 5C-8. Tampa Bay seagrass and marsh restoration projects.

For more information on this project, contact:

Tampa Bay Regional Planning Council
Agency on Bay Management
9455 Koger Boulevard
Tampa, Florida 33702

Salmon River Marsh Restoration Project

Approximately 75 percent of the Salmon River estuarine marshes in Lincoln County, Oregon, have been altered by dikes designed to isolate these wetlands from estuarine circulation and produce pastureland (Frenkel and Morlan 1991). Most of the original marshlands had been grazed by cattle or harvested for hay before dike construction began in the early 1960s (Mitchell 1981). According to Frenkel and Morlan (1991), these salt marshes were seldom flooded by salt water in summer and were deeply dissected by tidal creeks. Construction of dikes eliminated tidal access to most of the marshes and caused indirect impacts on adjoining, undiked marshland.

Restoration of the Salmon River impounded marshes (Figure 5C-9) began in 1978, when the U.S. Forest Service acquired approximately 80 acres of pasture and adjacent high salt marsh. The objectives of this restoration project were to reestablish estuarine circulation in the diked marshlands and return the estuary to its condition prior to agricultural use.

Restoration Approach

A dike that impounded 52 acres of salt marsh along the north side of the estuary was removed in 1978. Breaching of the dike system resulted in rapid restoration of hydrologic function, and vegetation changed quickly from pasture to salt marsh species. Subsidence of the ground surface associated with diking (on the order of 0.3 m) was beginning to be compensated for by sediment deposition (locally up to 0.07 m). Large barren areas were revegetated by marsh species by 1984. By 1988, intertidal marsh communities predominated in more than 90 percent of restored plots.

Physical and chemical parameters were also monitored, including site elevation, salinity, sediment texture and organic content, and accretion.

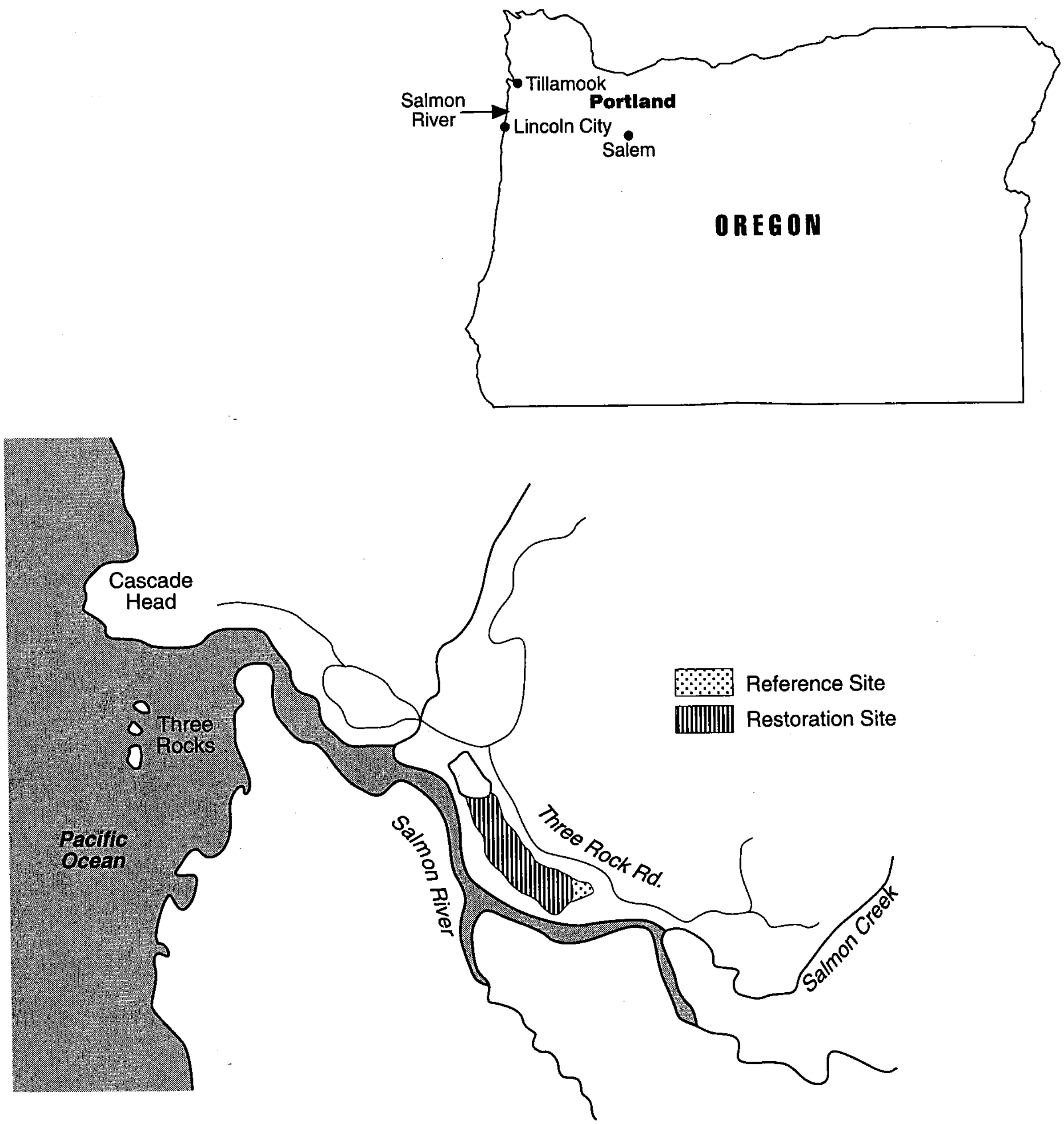


FIGURE 5C-9. Salmon River salt marsh restoration project.

Evaluation of Restoration Efforts

The project had qualified success because the restored marsh did not represent the high salt marsh community type that had been present prior to diking. Instead, low salt marsh vegetation recolonized the site. Natural sedimentation processes are likely to gradually allow vegetation associated with high marsh locations to recover. Although consideration was given to additional dike removal, further adjustments to the system were not implemented; rather, the re-opened marsh was allowed to attain a level of restoration that was productive and represented an improved, but not pre-disturbance, condition.

For more information on this project, contact:

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REFERENCES

Bell, S.S. 1979. Short- and long-term variation in a high marsh meiofaunal community. Est. Coast. Shelf Sci. 9:331–350.

Bittaker, H.F. 1975. A comparative study of the phytoplankton and benthic macrophyte primary productivity in a polluted versus an unpolluted coastal area. Thesis. Florida State University, Tallahassee, FL.

Broome, S.W. 1990. Creation and restoration of tidal wetlands of the southeastern United States. pp. 37–72. In: Wetland Creation and Restoration: The Status of the Science. J.A. Kusler and M.E. Kentula (eds). Island Press, Washington, DC.

Brown Jr., L.F., J.L. Brewton, J.H. McGowen, T.J. Evans, W.L. Fisher, and C.G. Groat. 1976. Environmental geologic atlas of the Texas coastal zone - Corpus Christi area. University of Texas at Austin, Bureau of Economic Geology, Austin, TX. 23 pp.

Clark, P.A. 1989. Seagrass restoration: a non-destructive approach. pp. 57–70. In: Proceedings of the 16th Annual Conference on Wetlands Restoration and Creation. May 25-26, 1989, Hillsborough Community College, Plant City, FL. F.J. Webb, Jr. (ed).

Clark, P.A. 1992. Tidal marsh restoration using volunteer fishermen. pp. 36–41. In: Proceedings of the 19th Annual Conference on Wetlands Restoration and Creation. May 14–15, 1992, Hillsborough Community College, Plant City, FL. F.J. Webb, Jr. (ed).

de la Cruz, A. 1979. Production and transport of detritus in wetlands. pp. 162–174. In: Wetland Functions and Values: The State of our Understanding. Proc. Nat. Sym. Wetlands. P. Greeson, J.R. Clark, and J.E. Clark (eds). American Water Resources Association, Minneapolis, MN.

den Hartog, C. 1970. The seagrasses of the world. North-Holland, Amsterdam. 275 pp.

Edmondson, J.B., and R.S. Jones. 1985. Barrier Island and Back Barrier marsh reconstruction in Terrebonne Parish, Louisiana. pp. 401–418. In: Proc. Second Water Quality and Wetlands Management Conference: Estuaries, New Orleans, LA, October 24–25, 1985. N.V. Brodtman (ed).

Eleuterius, L.N. 1976. The distribution of *Juncus roemerianus* in salt marshes of North America. Chesapeake Sci. 17:289–292.

Estevez, E.D., and L. Mosura. 1985. Emergent vegetation. pp. 248–278. In: Proceedings Tampa Bay Area Scientific Information Symposium. Florida Sea Grant College, Report No. 65. Treat et al. (eds).

Findlay, S., K. Schoeberl, and B. Wagner. 1989. Abundance, composition, and dynamics of the invertebrate fauna of a tidal freshwater wetland. J. N. Am. Benthol. Soc. 8:140–148.

Fonseca, M.S. 1990. Regional analysis of the creation and restoration of seagrass systems. pp. 171–193. In: Wetland Creation and Restoration: The Status of the Science. J.A. Kusler and M.E. Kentula (eds). Island Press, Washington, DC.

Frenkel, R.E., and J.C. Morlan. 1991. Can we restore our salt marshes? Lessons from the Salmon River, Oregon. Northwest Environ. J. 7:119–135.

Green, K.A. 1978. A conceptual model for Chesapeake Bay. FWS/OBS-79/69. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC.

Heck, K.L., Jr., K.W. Able, M.P. Fahay, and C.T. Roman. 1989. Fishes and decapod crustaceans of cape cod eelgrass meadows: species composition, seasonal abundance patterns and comparison with unvegetated substrates. Estuaries 12(2):59–65.

Iverson, R.L., and H.F. Bittaker. 1986. Seagrass distribution and abundance in eastern Gulf of Mexico coastal waters. Est. Coast. Shelf Sci. 22:577–602.

Jordan, T.E., J.W. Pierce, and D.L. Correll. 1986. Flux of particulate matter in the tidal marshes and subtidal shallows of the Rhode River estuary. Estuaries 9(4b):310–319.

Josselyn, M., M. Fonseca, T. Neisen, and R. Larson. 1986. Biomass, production, and decomposition of a deepwater seagrass, *Halophila decipiens* Ostenfeld. Aquat. Bot. 25:47–61.

Kadlec, R.H., and J.A. Kadlec. 1979. Wetlands and water quality. pp. 436–456. In: Wetland Functions and Values: The State of our Understanding. Proc. Nat. Sym. Wetlands. P. Greeson, J.R. Clark, and J.E. Clark (eds). American Water Resources Association, Minneapolis, MN.

Kneib, R.T. 1984a. Patterns in the utilization of the intertidal salt marsh by larvae and juveniles of *Fundulus heteroclitus* (Linnaeus) and *Fundulus luciae* (Baird). J. Exp. Mar. Biol. Ecol. 83:41–51.

Kneib, R.T. 1984b. Patterns of invertebrate distribution and abundance in the intertidal salt marsh. Causes and questions. Estuaries 7(4a):392–412.

Kneib, R.T. 1987. Seasonal abundance, distribution and growth of postlarval and juvenile grass shrimp (*Palaemonetes pugio*) in a Georgia, USA, salt marsh. Mar. Biol. 96:215–223.

Knutson, P.L., and W.W. Woodhouse, Jr. 1982. Pacific coastal marshes. pp. 111–130. In: Creation and Restoration of Coastal Plant Communities. R.R. Lewis, III (ed). CRC Press, Boca Raton, FL.

Kruczynski, W.L. 1982. Saltmarshes of the northeastern Gulf of Mexico. pp. 71–87. In: Creation and Restoration of Coastal Plant Communities. R.R. Lewis, III (ed). CRC Press, Boca Raton, FL.

Leber, K.M. 1985. The influence of predatory decapods, refuge and microhabitat selection on seagrass communities. Ecology 66(6):151–164.

Lewis, F.G. 1984. Distribution of macroinvertebrate crustaceans associated with *Thalassia, Halodule*, and bare sand substrata. Mar. Ecol. Prog. Ser. 19:101–113.

Lewis, F.G., and A.W. Stoner. 1983. Distribution of macrofauna within seagrass beds: an explanation for patterns of abundance. Bull. Mar. Sci. 33:296–304.

Lewis, R.R., III. 1982. Low marshes, peninsular Florida. pp. 47–152. In: Creation and Restoration of Coastal Plant Communities. R.R. Lewis, III (ed). CRC Press, Boca Raton, FL.

Lewis, R.R., III. 1987. The restoration and creation of seagrass meadows in the southeast United States. pp. 153–177. In: Proceedings of the Symposium on Subtropical-Tropical Seagrasses of the Southeastern United States. M.J. Durako, R.C. Phillips, and R.R. Lewis III (eds). Florida Marine Research Publications 42.

Lewis, R.R., III, R.G. Gilmore, Jr., D.W. Crewz, and W.E. Odum. 1985a. Mangrove habitat and fishery resources of Florida. pp. 281–336. In: Florida Aquatic Habitat and Fishery Resources. W. Seaman, Jr. (ed). Florida Chapter of the American Fisheries Society, Kissimmee, FL. 543 pp.

Lewis, R.R., III, M.J. Durako, M.D. Moffler, and R.C. Phillips. 1985b. Seagrass meadows of Tampa Bay: a review. pp. 210–246. In: Proceedings Tampa Bay Area Scientific Information Symposium. Florida Sea Grant College, Report No. 65. Treat et al. (eds).

Linsey, D.W., and A.V. Linsey. 1972. Alabama wildlife: amphibians, reptiles, birds, and mammals. Linsey and Linsey, Mobile, AL. 109 pp.

Lombardo, R., and R.R. Lewis III. 1985. Commercial fisheries data: Tampa Bay. pp. 614–634. In: Proceedings Tampa Bay Area Scientific Information Symposium. Florida Sea Grant College, Report No. 65. Treat et al. (eds).

Los Peñasquitos Lagoon Foundation and State Coastal Conservancy. 1985. Los Peñasquitos Lagoon enhancement plan and program. State Coastal Conservancy, Oakland, CA.

Marcus, L. 1989. The coastal wetlands of San Diego County. State Coastal Conservancy, Oakland, CA.

McConnaughey, B.H., and R. Zottoli. 1983. Introduction to marine biology. D. Bowen (ed). C.V. Mosby Co., St. Louis, MO. 638 pp.

McIvor, C.C., and W.E. Odum. 1988. Food, predation risk, and microhabitat selection in a marsh fish assemblage. Ecology 69:1341–1351.

Mendelssohn, I.A., K.L. McKee, and W.H. Patrick, Jr. 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaptation to anoxia. Science 214:439–441.

Mitchell, D.L. 1981. Salt marsh reestablishment following dike breaching in the Salmon River Estuary, Oregon. Dissertation. Oregon State University, Corvallis, OR.

Mitsch, W.J., and J.G. Gosselink. 1993. Wetlands. Second Edition. Van Nostrand Reinhold, New York, NY.

Nixon, S.W. 1982. The ecology of New England high salt marshes: a community profile. FWS/OBS-81/55. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 70 pp.

Odum, W.E. 1988. Comparative ecology of tidal freshwater and salt marshes. Ann. Rev. Ecol. Sys. 19:147–176.

Odum, W.E., and E.J. Heald. 1972. Trophic analyses of an estuarine mangrove community. Bull. Mar. Sci. 22(3):671-738.

Odum, W.E., and M.A. Heywood. 1978. Decomposition of intertidal freshwater marsh plants. pp. 89-97. In: Freshwater Wetlands. R.E. Good, D.F. Wingham, and R.L. Simpson (eds). Academic Press, New York, NY.

Odum, W.E., M.L. Dunn, and T.J. Smith. 1979. Habitat value of tidal freshwater wetlands. pp. 248-255. In: Wetland Functions and Values: The State of our Understanding. Proc. Nat. Sym. Wetlands. P. Greeson, J.R. Clark, and J.E. Clark (eds). American Water Resources Association, Minneapolis, MN.

Odum, W.E., T.J. Smith, J.K. Hoover, and C.C. McIvor. 1984. The ecology of tidal freshwater marshes of the United States East Coast: a community profile. FWS/OBS-83/17. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. 177 pp.

Orth, R.J. 1977. The importance of sediment stability in seagrass communities. pp. 281-300. In: Ecology of Marine Benthos. B.C. Coull (ed). University of South Carolina Press, Columbia, SC.

Orth, R.J., and K.L. Heck, Jr. 1980. Structural components of eelgrass (*Zostera marina*) meadows in the lower Chesapeake Bay- Fishes. Estuaries 3(4):278-288.

Orth, R.J., K.L. Heck, Jr., and J. van Montfrans. 1984. Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships. Estuaries 7(4a):339-350.

Penland, S., and R. Boyd. 1981. Shoreline changes on the Louisiana barrier coast. Oceans, September 1981, pp. 209-219.

Peterson, C.H. 1982. Clam predation by whelks (*Busycon* spp.): experimental tests of the importance of prey size, prey density, and seagrass cover. Mar. Biol. 66:159-170.

Phillips, R.C. 1980. Responses of transplanted and indigenous *Thalassia testudinum* Banks ex Konig and *Halodule wrightii* Ascherson to sediment loading and cold stress. Contrib. Mar. Sci. 23:79-87.

Phillips, R.C. 1982. Seagrass meadows. pp. 173-201. In: Creation and Restoration of Coastal Plant Communities. R.R. Lewis, III (ed). CRC Press, Boca Raton, FL.

Prestegaard, K. 1978. Stream and lagoon channel of the Los Peñasquitos Watershed, California, with an evaluation of possible effects of proposed urbanization. California Coastal Commission Study Series.

Proctor, C.M., J.C. Carcia, D.V. Galvin, G.B. Lewis, and L.C. Loehr. 1980. An ecological characterization of the Pacific Northwest Coastal region. Volume 3: Characterization atlas-zone and habitat descriptions. FWS/OBS-79/13. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC.

Rader, D.N. 1984. Salt marsh benthic invertebrates: small scale patterns of distribution and abundance. Estuaries 7(4a):413–420.

Reidenbaugh, T.G., and W.C. Banta. 1980. Origin and effects of *Spartina* wrack in a Virginia salt marsh. Gulf Res. Repts. 6:393–401.

Rejmankova, E. 1992. Ecology of creeping macrophytes with special reference to *Ludwigia peploides* (H.B.K.) Raven. Aquat. Bot. 43:283–299.

Richardson, C.J. 1979. Primary productivity in freshwater wetlands. pp. 131–145. In: Wetland Functions and Values: The State of our Understanding. Proc. Nat. Sym. Wetlands. P. Greeson, J.R. Clark, and J.E. Clark (eds). American Water Resources Association, Minneapolis, MN.

Rozas, L.P., and W.E. Odum. 1987. Fish and macrocrustacean use of submerged plant beds in tidal freshwater marsh creeks. Mar. Ecol. Prog. Ser. 38:101–108.

Rozas, L.P., and M.W. Lasalle. 1990. A comparison of the diets of Gulf killifish, *Fundulus grandis* Baird and Girard, entering and leaving a Mississippi marsh. Estuaries 13:332–336.

SANDAG. 1982. Los Peñasquitos Lagoon watershed management plan. Prepared for San Diego Association of Governments, San Diego, CA. Boyle Engineering, Newport Beach, CA.

Schomer, N.S., and R.D. Drew. 1982. An ecological characterization of the lower Everglades, Florida Bay, and the Florida Keys. FWS/OBS-82/58. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC.

Schubauer, J.P., and C.S. Hopkinson. 1984. Above- and belowground emergent macrophyte production and turnover in a coastal marsh ecosystem, Georgia. Limnol. Oceanogr. 29:1052–1065.

Silberhorn, G.M. 1976. Tidal wetland plants of Virginia. Virginia Institute of Marine Science. Gloucester Point, VA.

Sinicrope, T.L., P.G. Hine, R.S. Warren, and W.A. Niering. 1990. Restoration of an impounded salt marsh in New England. Estuaries 13(1):25–30.

Stone, J.H., L.M. Barr, Jr., and J. Day. 1978. Effects of canals on freshwater marshes in coastal Louisiana and implications for management. pp. 299–320. In: Freshwater Wetlands: Ecological Processes and Management Potential. R.E. Good, D.F. Whigham, and R.L. Simpson (eds). Academic Press, New York, NY.

Stoner, A.W. 1980. The role of seagrass biomass in the organization of benthic macrofaunal assemblages. Bull. Mar. Sci. 30(3):537–551.

Suchanek, T.H. 1983. Control of seagrass communities and sediment distribution by *Ceylonese* (Crustacea, Thalassinidea) bioturbation. J. Mar. Res. 41:281–298.

Summerson, H.C., and C.H. Peterson. 1984. Role of predation in organizing benthic communities of a temperate-zone seagrassbed. Mar. Ecol. Prog. Ser. 15:63–77.

Taylor, M.H., and L. DiMichelle. 1983. Spawning site utilization in a Delaware population of *Fundulus heteroclitus* (Pices: Cyprinodontidae). Copeia 1983(3):719–725.

Thayer, G.W., H.H. Stuart, W.J. Kenworthy, J.F. Ustach, and A.B. Hall. 1979. Habitat values of salt marshes, mangroves, and seagrasses for aquatic organisms. pp. 235–247. In: Wetland Functions and Values: the State of our Understanding, Proc. Nat. Sym. Wetlands. P. Creeson, J.R. Clark, and J.E. Clark (eds). American Water Resources Association, Minneapolis, MN.

Thayer, G.W., D.R. Colby, and W.F. Hettler, Jr. 1987. Utilization of the red mangrove prop root habitat by fishes in south Florida. Mar. Ecol. Prog. Ser. 35:25–38.

Thibaut, T.D. 1992. Factors affecting the distribution and abundance of the big-clawed snapping shrimp *Alpheus heterochaelis* (Say) within an Alabama shoal-grass meadow. Thesis. Auburn University, AL. 79 pp.

Thomas, J.L., R.J. Zimmerman, and T.J. Minello. 1990. Abundance patterns of juvenile blue crabs (*Callinectes sapidus*) in nursery habitats of two Texas bays. Bull. Mar. Sci. 46(1):115–125.

Tiner, R.W. 1993. Coastal wetland plants of the Southeastern United States. University of Massachusetts, Amherst, MA.

Treat, S.F., J.L. Simon, R.R. Lewis, III, and R.L. Whitman, Jr. 1985. Proc. Tampa Bay Area Scientific Information Symposium. Florida Sea Grant College, Report No. 65.

Van Montfrans, J., R.L. Wetzel, and R.J. Orth. 1984. Epiphyte-grazer relationships in seagrass meadows: consequences for seagrass growth and production. Estuaries 7(4a):289–309.

Virnstein, R.W., P.S. Mikkelson, D.D. Cairns, and M. Capone. 1983. Seagrass beds versus sand bottoms: the trophic importance of their associated benthic invertebrates. Fla. Sci. 45:363–381.

Ward, L.G., and R.R. Twilley. 1986. Seasonal distributions of suspended particulate material and dissolved nutrients in a coastal plain estuary. Estuaries 9(3):156–158.

Wass, M.L., and T.D. Wright. 1969. Coastal wetlands of Virginia: a summary of the interim report to the Governor and General Assembly. Virginia Institute of Marine Science, Gloucester Point, VA.

Watzin, M.C., and J.G. Gosselink. 1992. The fragile fringe: coastal wetlands of the continental United States. U.S. Fish and Wildlife Service, Washington, DC.

Webb Jr., J.W. 1982. Salt marshes of the western Gulf of Mexico. pp. 89–109. In: Creation and Restoration of Coastal Plant Communities. R.R. Lewis, III (ed). CRC Press, Boca Raton, FL.

Williams, A.H., L. Cohen, and M. Stoelting. 1990. Seasonal abundance, distribution, and habitat selection of juvenile *Callinectes sapidus* Rathbun in the northern Gulf of Mexico. J. Exp. Mar. Biol. Ecol. 137:165–183.

Williams, G.D., and D.R. Gibson. 1995. The physical, chemical, and biological monitoring of Los Peñasquitos Lagoon, 20 December 1994 - 20 March 1995. Quarterly report to Los Peñasquitos Lagoon Foundation.

Woodhouse Jr., W.W., and P.L. Knutson. 1982. Atlantic coast marshes. pp. 45–70. In: Creation and Restoration of Coastal Plant Communities. R.R. Lewis, III (ed). CRC Press, Boca Raton, FL.

Yozzo, D.J., and D.E. Smith. 1995. Abundance, microhabitat distribution, and seasonability of meiofauna from a Chickahominy River, Virginia tidal freshwater marsh. Hydrobiologia 310:197–206.

Zieman, J.C. 1976. The ecological effects of physical damage from motor boats on turtle grass beds in southern Florida. Aquat. Bot. 2:127–139.

Zieman, J.C. 1987. A review of certain aspects of the life, death, and distribution of the seagrasses of the southeastern United States, 1960–1985. Fla. Mar. Res. Publ. No. 42:53–76.

Zimmerman, R.J., and T.J. Minello. 1984. Densities of *Penaeus aztecus, Penaeus setiferus,* and other natant macrofauna in a Texas salt marsh. Estuaries 7(4a):421–433.

5D. FRESHWATER WETLANDS

Daniel Willard and Anne MacDonald

Freshwater (palustrine) wetlands are the most widely distributed of aquatic habitats, occurring from the headwaters to the mouths of great river systems and from drainage divides to river valleys. They are defined as vegetated (greater than 30 percent cover), are often small (if open water, less than 8 ha), and have shallow water (less than 2 m deep) areas (Cowardin et al. 1979). Freshwater wetlands may have herbaceous, shrubby, or forested vegetation. They may be temporarily flooded, saturated, or continually flooded, and their hydrology changes from season to season. These wetlands occur from subtropical to arctic regions from sea level to high altitudes. Because of this wide geographic and ecological distribution and because wetlands are transitional between dry land and open water, they represent a wide variety of habitat.

Historically, the ecosystem value of wetlands has competed poorly with the potential commercial value of these lands in a drained condition. Only in the last 20 years have regulatory actions been taken to protect wetlands from severe anthropogenic disturbance. Even when these areas are temporarily denuded of natural vegetation or soil by floods, hurricanes, or anthropogenic activities, they will continue to function as wetlands over the long term. Therefore, this discussion is not limited to just "jurisdictional" wetlands, as defined by either federal or state regulation or guidance, but also pertains to "wet lands" not dealt with elsewhere in this document.

Change is a natural occurrence in wetlands. Water levels fluctuate, often in a cyclic way (Figure 5D-1). Rivers change course. Lake littoral zones expand and contract. Unpredictable perturbations (i.e., "disturbance") reset successional clocks. The fact that change is a natural condition shapes all delineation, regulation, or management decisions. Scientists must describe wetlands in conditional terms, necessarily stating their conclusions with spatial and temporal modifiers such as "the water table is *usually* above the surface a few months of the year," "this is a *50-year* floodplain," "summer drought *often occurs* in July or August," or "northern white cedar occur over a wide range of mesic conditions, but *seem to reach dominance* in two ecotypes, a moist variety and a more dry-loving group." This section will illuminate the significant role that disturbance has in a wetland system and the implications of disturbance for wetland restoration.

Included in this section is a discussion of classification schemes for freshwater wetlands, as they have developed over many years of wetlands management and regulation. While understanding many of these classification schemes is not a

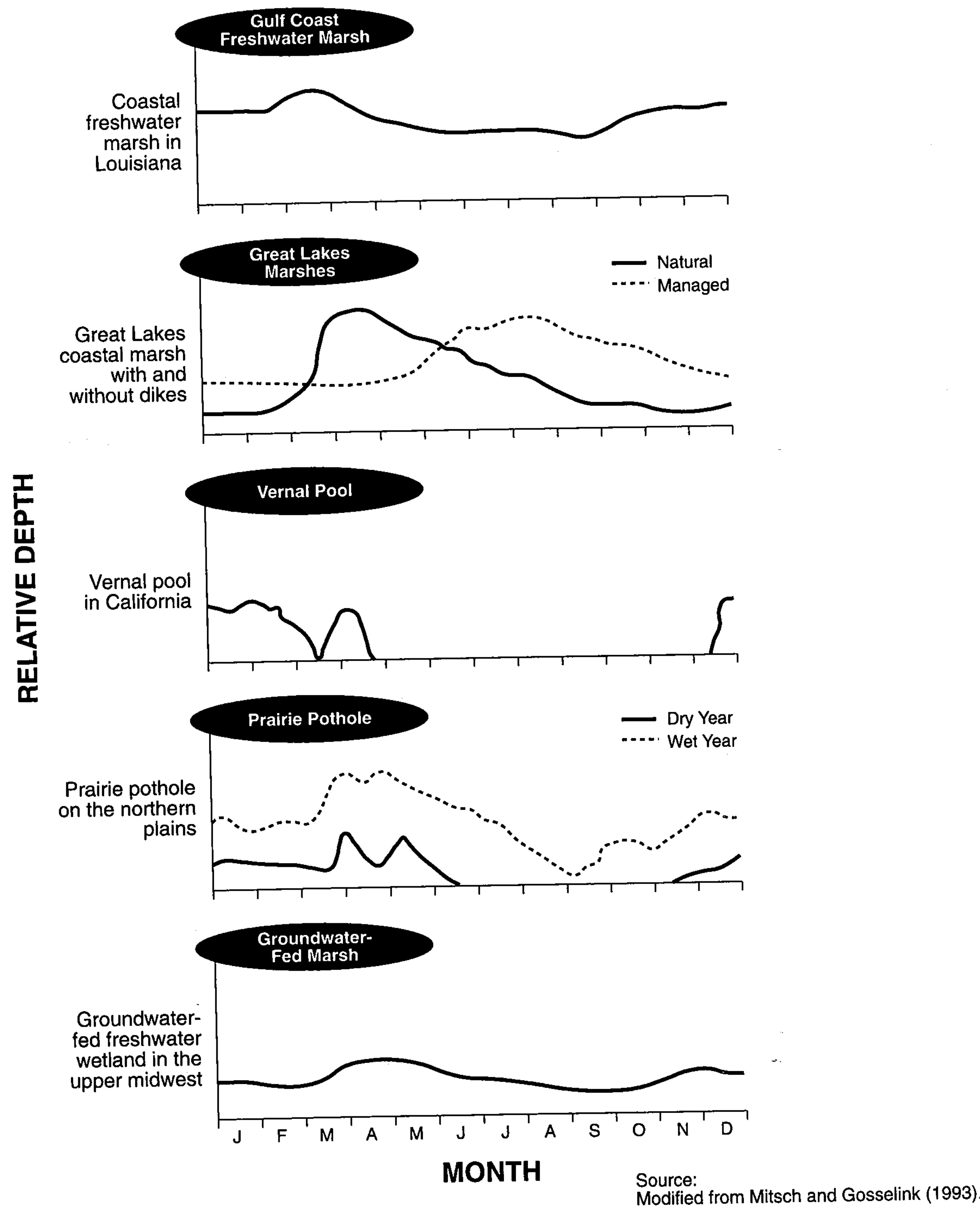


FIGURE 5D-1. Typical hydroperiods for various freshwater marshes.

prerequisite to restoration, it is a necessary part of using historical and synoptic information that may be available for a site or region. An exception to this is the hydrogeomorphic (HGM) method developed by Brinson (1993a), because an understanding of site water sources allows one to make judgments concerning attributes of the key environmental parameter at a site: how wet for how long, over what spatial scale. The HGM method also provides a way to extrapolate information on wetland functions from reference wetlands in a region to the site at hand.

Freshwater wetlands may be isolated features surrounded by uplands, in riparian and floodplain settings, in the freshwater tidally influenced portion of estuaries, and fringing lakes (including freshwater "estuaries" in the Great Lakes). Fluctuations in water levels, transport and removal of nutrients, detritus, and sediments in wetlands are often controlled by these adjoining ecosystems. In addition, restoration of fringing wetlands may contribute to the restoration of the larger water body, as demonstrated in the Lake Apopka case study presented in the *Lakes and Reservoirs* section and discussed by Mitsch (1995). Therefore, readers with projects near any of these other ecosystems are urged to also consult the other appropriate ecosystem sections within the *Ecosystem and Restoration Profiles* chapter.

More detail on the processes and characteristics of wetlands discussed in this section can be found in Mitsch and Gosselink (1993). This reference is currently the most comprehensive discussion of freshwater wetlands available. In addition to text, it contains an extensive bibliography of references dating through 1992 and a detailed index that includes common and scientific names for wetland flora and fauna.

WETLAND CLASSIFICATION SYSTEMS

Most wetland experts are familiar with Habitat Evaluation Procedures (HEP) (USFWS 1980), Wetland Evaluation Technique (WET) (Adamus et al. 1987), the various interagency classification manuals (e.g., Corps 1987), Circular 39 (Shaw and Fredine 1956), and the new HGM (Brinson 1993a), and use them interchangeably as the need arises. Classification systems and evaluation models all reflect the purpose and training of their creators. Regardless of the classification scheme(s) used, an understanding of wetlands must accommodate three issues. First, the confounding problem with comparing ecosystems arises from the contrast between these two statements:

- Each ecosystem is unique
- Much of our traditional scientific knowledge about ecosystems depends on theory and generalization.

Therefore, while the patterns and processes that exist at one place resemble those at another, the actual consequences of the local interaction may vary considerably from place to place. Natural history studies act to validate and calibrate the local applicability of the deductions from traditional science.

Second, water levels and patterns of vegetation and habitat use fluctuate within certain temporal and spatial ranges. Wetlands are, by their very nature, shallow water and high groundwater systems. This combination makes them different from either water or land and gives them some special qualities. Many wetland functions are dependant on the water level fluctuations. In contrast, natural fluctuations in water levels that result from seasonal or long-term precipitation cycles do not dramatically change the appearance or boundaries of lakes, streams, rivers, or oceans. But as shallow surface water and high groundwater systems with gentle slopes, wetland boundaries are often difficult to locate precisely. Differences in water levels of inches, caused by normal fluctuations in precipitation or watershed activities, may make the difference between "wetland" and "nonwetland" or dramatically change wetland plant species. Over very short periods of time (measured in weeks to months), "wetlands" can undergo intense selection pressure on the resident plants and animals.

Yet, wetlands do have permanence in the landscape and relatively certain boundaries when viewed from the long-term perspective. The fluctuations occur within relatively fixed limits. Because precipitation varies seasonally, annually, and over long-term cycles, a prairie pothole or other wetland may be wet year-round for 2 years, seasonally for the next 5 years, and then almost entirely dry for the following 5 years (Figure 5D-1). The recent drought in the western United States has demonstrated that the critical feeding and resting values of wetlands for ducks, geese, and other waterfowl depends on seasonal wetness and wetness in the "dry years" as well. The key to understanding frequency of inundation is that it is not an absolute annual, every-other-year, or every-3-year event. It is a periodic event with a range of hydrologic conditions that occur within a given region or watershed of the country. Therefore, both a long- and short-term perspective of hydrology is required.

Third, fluctuations in water levels result in a combination of natural functions and natural hazards that are not readily apparent upon a casual site visit, particularly during dry periods. Wetlands are not static or relatively static systems that can be delineated or classified based on a single determination of existing water level or vegetation. A "one-shot" view of wetlands based on a single field examination of wetland hydrology at the time of a site visit cannot reflect values and functions, nor can it accurately reflect the hydrologic or other wetland characteristics. The relatively hidden nature of these functions and values and the costs of documenting the functions and values are two of the reasons wetland evaluation, classification, regulation, and restoration are so difficult, time-consuming, and expensive.

To make this section consistent with both current and older wetland literature, the wetland classification systems of both HGM and Cowardin et al. (1979) are used to assist in understanding wetland ecosystems. These schemes are complimentary but not directly linked. HGM is descriptive of the geophysical properties of wetlands, while the Cowardin et al. (1979) scheme describes floral structure as a function of gross setting (marine vs. riverine) and water regime. Table 5D-1 provides a comparison of Cowardin et al. (1979) with the earlier Circular 39 (Shaw and Fredine 1956) classification and with the National Wetland Inventory and HEP cover types.

Hydrogeomorphic Approach to Wetlands Analysis

For purposes of assessing and restoring wetland processes and functions relative to a larger landscape, HGM is a useful starting point. HGM assumes that knowledge of the geomorphic position of the wetland, understanding of the water source(s), and estimates of hydrodynamics (all of which interact in a limited number of ways) can provide significant clues to the ecosystem functions and values of the site (these functions and values will be discussed in greater detail later in this section). Observations necessary to classify a wetland using the HGM system can be made at any time of the year by an experienced hydrologist or geomorphologist because these one-time observations are implicitly put in a longer perspective (seasonal, annual, or multiyear) by one with this experience.

For instance, one of the geomorphic attributes used by HGM is the relative position of a site in the drainage basin. If a wetland is riparian and near the headwaters of a rain-fed stream, water levels will respond rapidly to flood discharges. Hydrographs in these settings tend to have high, sharp peaks because routing times are short and flood peaks are approximately in-phase in the upper portions of drainage basins. Therefore, periods of inundation are temporally limited. In contrast, riparian wetlands low in the drainage basin (e.g., the lower reaches of major river systems) will experience lower flood peaks of long duration and an extended period of saturated soils. The combination of geomorphic position, water source (both upstream inlet and downstream outlets of surface water; Figure 5D-2), and associated hydrodynamics completely control water quantity and quality functions of the wetlands and significantly influence the more complex biotic functions. Tables 5D-2, 5D-3, and 5D-4, respectively, demonstrate how observations about the geomorphic setting, water source and climate, and hydrodynamics can be used to infer function and ecological significance of a particular site. Brinson (1993b) further summarizes the degree to which wetland function changes along HGM gradients.

TABLE 5D-1. COMPARISON OF WETLAND CLASSIFICATION SYSTEMS

A. Comparison of Cowardin et al. (1979) and Circular 39 (Shaw and Fredine 1956)

Cowardin et al. (1979)	Circular 39 (Shaw and Fredine 1956)
Aquatic Bed	Types 4 and 5: deep fresh marshes and open freshwater
Emergent Wetland	Types 1, 2 and 3: seasonally flooded basins or flats, fresh meadows, and shallow fresh marshes
Scrub–Shrub Wetland	Shrub swamps
Forested Wetland	Wooded swamps

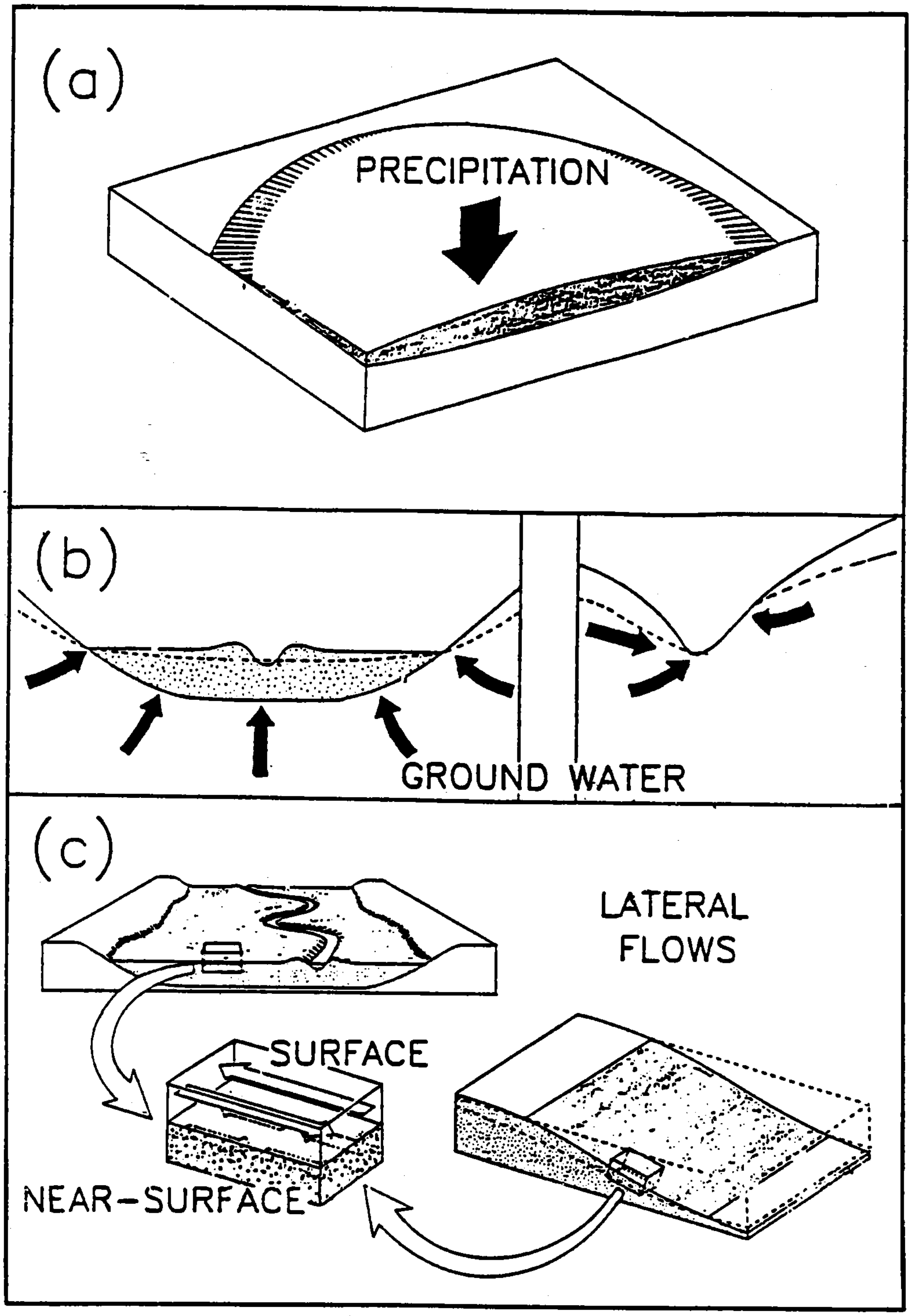
B. Comparison of Cowardin et al. (1979), National Wetland Inventory (NWI), and Habitat Evaluation Procedure (HEP) Cover Types

Aquatic Bed is noted on NWI maps as AQ, but is often not listed. In HEP models, aquatic bed is covered as shallow water or littoral zone with variables describing vegetation.

Emergent Wetland is noted on NWI maps as Palustrine Emergent Marsh (PEM) with a variety of modifiers (see Cowardin et al. [1979] for modifiers). HEP has changed over the years. Formerly this was Herbaceous Wetland (HW) and sometimes Emergent Wetland (EW).

Scrub–shrub is Palustrine scrub shrub [Pss] on NWI maps. HEP often lists it as Deciduous Scrub Wetland [DSW], with modifiers as above.

Forested Wetland is Palustrine Forested Wetland with several classes such as deciduous or evergreen, broad-leaved or needle-leaved. NWI lists these wetlands with a class and usually a modifier.



Source: Brinson (1993a).

FIGURE 5D-2. Principal sources of water.

TABLE 5D-2. EXAMPLES OF GEOMORPHIC SETTING AS A PROPERTY OF HYDROGEOMORPHIC CLASSIFICATION

Examples of Geomorphic Setting[a]	Qualitative Evidence[b]	Quantitative Evidence[c]	Functions[d]	Ecological Significance[e]
Depressional Wetlands[f]				
No apparent inlet or outlet.	Topographically isolated from other surface water bodies.	Drydowns frequent; water table significantly below wetland much of the time.	Retains inflow; loss primarily by evapotranspiration or infiltration.	Inaccessible to aquatic organisms dependent on streams. Endemism likely.[g]
Positioned on local topographic high. Surface outlet only.	Outlet may be defined by contours or intermittent stream symbol.	Drydowns frequent; water table significantly below wetland much of the time.	Temporary flood storage; outlet may overflow during high water (surface-water dominated) or flow continuously (groundwater supported). Outlet controls maximum depth.	System open to upstream immigration and downstream emigration of aquatic organisms. Potential for recolonization by aquatic organisms if drydowns cause local extinctions.
Located in marginally dry climate (e.g., prairie pothole region). Variable inlets and outlets.	Inlets and outlets may be defined by contours or intermittent stream symbol.	If water has low conductivity, wetland is recharging underlying aquifer. If high conductivity, groundwater is discharging to wetland (Sloan 1972).	Retains inflow; loss primarily by evapotranspiration or infiltration. May be subject to wide fluctuation in water depth.	Geographic location critical to migrating waterfowl as flyway position indicates. Changes in vegetation create varied waterfowl habitat. May be vulnerable to eutrophication and toxin accumulation because of long residence time of water. Probably import and export of detritus.
Both surface inlet and outlet; large catchment sustains marginal riverine features.	Inlets and outlets may be defined by contours or intermittent stream symbol.	Water budget dominated by lateral surface flows or strong groundwater discharge.	Temporary flood storage; drainage back to stream shortly after flooding (surface-water-supported) or continuous saturation (groundwater-supported).	Potential for fish population recruitment through migration. Probably import and export of detritus.
Located on break in slope.	Soil saturated most of time.	Chemistry indicative of groundwater; discharge from slope base or face. Piezometric confirmation.	Inflow steady and continuous; loss by evapotranspiration seasonal. Renewal of pore waters maintains higher redox than typical for constant saturation. Low surface storage capacity.	Provides surface moisture during dry periods; contributes to beta (i.e., regional) diversity of landscape.

TABLE 5D-2. (cont.)

Examples of Geomorphic Setting[a]	Qualitative Evidence[b]	Quantitative Evidence[c]	Functions[d]	Ecological Significance[e]
Extensive Peatland				
Ombrotrophic bog.	Peat substrate; saturated most of time. Plant species indicate ombrotrophic bog; surface flows negligible.	Peat confirmed by organic content and thickness. Ombrotrophy evident from low pH and low ion content.	Surface storage may facilitate storm runoff; groundwater conservation occurs when water table is below surface. Peat deposits control topography and geomorphic surface.	Wetland-upland interactions minor relative to wetland-atmospheric exchanges.[h] Upland habitats scarce. Species composition unique to bog conditions.
Rich fen.	Peat substrate; saturated most of time. Graminoid species indicative of groundwater supply.	Peat confirmed by organic content and thickness. Minerotrophy evident from circumneutral pH and high ion content.	Subsurface water supply maintains saturation to surface and hydraulic gradient to maintain flow.	Represents conduit for lateral water movement without channelized flow. Moderate levels of primary production and organic matter export.
Riverine Wetlands (floodplain, not channel)				
Streamside zones of intermittent streams.	Headwater position; first order stream.	Flows not continuous; flow lacks headwater flooding and overbank flow properties.	Interface of landscape where groundwater and surface water sources change phases to fluvial environment.	Riparian zone critical to maintaining buffer between upland and stream flow.
High-gradient:				
Downcutting portions[i]	Bedrock-controlled channel.	Substrate lacks alluvium (soil maps). Flow may be continuous but likely flashy.	Scour precludes extensive wetland development. Unvegetated reaches allow light penetration to support aquatic production.	May impede wildlife movement and cover if corridor too narrow. Maintains important in-stream riffle habitat.
Aggrading portions	Substrate controlled by fluvial processes.	Stratigraphy shows interbedding and coarse particle size (gravel and larger).	Wetland on coarse substrate maintained by upslope groundwater source.[j]	Unstable substrate in high-energy environment colonized by pioneer species. Streamside vegetation contributes to allochthonous organic supply.
Middle-gradient landform.	Channelized flow, evidence of oxbows, meander scrolls, etc., consistent with fluvial processes.	Flow likely continuous with moderate- to high-base flows.	Channel processes establish variation in topography, hydroperiod, and habitat interspersion on a floodplain.	Alluvium is renewed by surface accretion and point bar deposition; interspersion of plant communities contributes to beta (i.e., regional) diversity.[k]

TABLE 5D-2. (cont.)

Examples of Geomorphic Setting[a]	Qualitative Evidence[b]	Quantitative Evidence[c]	Functions[d]	Ecological Significance[e]
Riverine Wetlands (floodplain, not channel) (cont.)				
Low-gradient alluvial. Floodplain of bottomland hardwood.	As above, but in low-gradient landform.	Flow continuous with cool season flooding. High-suspended sediments in stream.	Flood storage; conserves groundwater discharge.	Major habitat for wildlife and biodiversity; strong biogeochemical activity and nutrient retention.
Low-gradient nonalluvial (i.e., low in suspended sediments): Florida cypress strands and sloughs, peat water tracks	Flows not channelized or channels shallow; if peatland, flow limited to acrotelm (upper 20 to 30 cm) (Clymo 1983, 1984).	Manning coefficient normally high.[l] Vegetation and sediment redox differ from surroundings if peatland.	Conduit for drainage in otherwise precipitation-dominated wetland. Flow facilitates nutrient availability.	Wetland possesses both depressional and riverine attributes because of weak lateral flows.
Fringe Wetlands				
Shoreline of large lake (i.e., lacustrine).	Subjected to seiches. Lake level controls position.	Amplitude and frequency of wind-generated fluctuations. Year-to-year trends in zonation follow climatic cycles.	Lake serves as water supply for wetland and establishes hydroperiod gradient for wetland zonation.	Provides shoreline stabilization under moderate wave action; transition habitat used by both aquatic and terrestrial organisms.
Coastal sea-level location (i.e., estuarine).	Subjected to astronomic tides; sea-level controlled.	Elevation relative to tides and changing sea level.	Wetland responsive to tides and changing sea level.	Barrier to saltwater encroachment; accommodates sediment deposition; open to estuarine organisms for feeding and recruitment.

Source: Brinson (1993a)

[a] Array of geomorphic settings is derived from the fringe-riverine-basin/depressional categories of Lugo et al. (1990). Extensive peatlands were originally not a separate category.

[b] Normally assessed by direct observation.

[c] Normally requires records of discharge and stage height to illustrate seasonal and interannual variation. Reliable indicators may be used also.

[d] Mechanisms for maintaining ecological significance.

[e] Other ecologically significant functions may be present; only examples are given.

[f] If wetland contains open water, waterfowl habitat may be inferred. For prairie pothole region, soil properties provide excellent indicators of hydrology (Hubbard 1988). Playa lakes of the southern High Plains reviewed by Bolen et al. (1989).

[g] Zedler (1987).

[h] Brinson (1991).

[i] Geomorphic processes and their ecological significance in riparian ecosystems are reviewed by Gregory et al. (1991).

[j] Major water supply to streamside wetlands is provided from upslope by groundwater discharge (Ruddy and Williams 1991).

[k] Metzler and Damman (1985) describe dependence of understory herbaceous vegetation on annual flood regime.

[l] Arcement and Schneider (1989).

TABLE 5D-3. EXAMPLES OF WATER SOURCE AND CLIMATE AS A PROPERTY OF HYDROGEOMORPHIC CLASSIFICATION

Examples of Water Sources[a] (and climatic setting)	Qualitative Scale[b]	Quantitative Estimate[c]	Functions[d]	Ecological Significance or Characters Maintained[e]
Precipitation (moist climate).	Precipitation dominates site water balance and water supply to plant community.[f]	Precipitation >PET during growing season.	Rarity of water table drawdown promotes organic matter accumulation, which further retards drainage; paludification is promoted.	Biogenic landscape isolates mineral soil from access by plants; low primary production eventually results.
Lateral surface or near-surface transport from overbank flow (mesic climate).	Discharge commonly exceeds bankfull-channel capacity.	Duration and frequency of overbank flow to floodplain can be inferred from hydrographs and floodplain elevation.	Overbank flow contributes to both flashy hydroperiod and vertical accretion of sediments. This creates rapid biogeochemical cycling and supplies nutrients.	Conditions maintained for high primary productivity and complex habitat structure.
Groundwater discharge to wetland (mesic climate).	Seeps occur at bases of hillslopes or below breaks in slope and along edges of streams and lakes.	Hydraulic gradient of groundwater increases with distance from wetland. Substrate permeable enough to allow flows.	Groundwater supplies nutrients, renews water, and flushes potential plant growth inhibitors.	Conditions conducive for stable plant community of high productivity. Peat accumulation possible
Both groundwater discharge and, during flood flows, lateral surface transport from upstream (arid climate).	Nonatmospheric sources greatly exceed supply from precipitation.	Precipitation <<PET during growing season.	High water tables are maintained by catchment supplies from upstream and from groundwater sources.	Water supplies support vegetative complexity and habitat structure not found in uplands because of water stress in arid climate.
All three sources, but precipitation is minor (subhumid to semiarid).	Alternate drought and wet periods produce decade-long cycles of water table fluctuations.	Precipitation <<PET.	High water levels induced by precipitation; groundwater discharge prevents extreme drawdowns; wetland may recharge groundwater when water table is high; conserves/reduces groundwater discharge when water levels are normal.	High primary productivity occurs when water is abundant; decomposition is rapid enough during dry periods to prevent peat accumulation.

Source: Brinson (1993a)

Note: PET - potential evapotranspiration

Footnotes on following page.

TABLE 5D-3. (cont.)

[a] Five simple examples with three climatic settings are represented by dominance of one type (first three) or mixtures of several sources of water illustrated in Figure 5D-2: precipitation, groundwater, and lateral surface or near-surface flow. In floodplains, groundwater is normally transported subterraneanly from the upland to the wetland. Surface water may be delivered by overbank flow from the stream channel or from channelized or nonchannelized flow by riparian transport.

[b] Climate and source of water are combined to illustrate four different types of wetland. Roughly speaking, the mesic climate has precipitation approximately equivalent to PET, the humid climate has precipitation >PET, and the arid climate has precipitation <<PET.

[c] Climatic records allow calculation of PET, which can be determined from the empirical formula of Holdridge et al. (1971): mean annual biotemperature × 58.93 = PET in millimeters, where mean annual biotemperature is the average of all values >0°C. Where precipitation falls short of PET, sources of water other than precipitation reduce site water deficits. Supplies from groundwater and inflows from upstream are necessary to maintain wetland conditions in all but moist climates.

[d] This column describes mechanisms by which hydrology contributes to and maintains wetland conditions.

[e] Other ecologically significant functions may exist; those listed serve only as examples.

[f] High amounts of orographic rainfall are necessary to maintain waterlogged conditions on sloped landscapes (Lugo et al. 1990).

TABLE 5D-4. EXAMPLES OF HYDRODYNAMIC PROPERTIES OF HYDROGEOMORPHIC CLASSIFICATION

Examples of Hydrodynamics[a]	Qualitative Evidence[b]	Quantitative Evidence[c]	Functions[d]	Ecological Significance[e]
Vertical Fluctuation of Water Table[f]				
Seasonal fluctuations nested within multiyear cycles.	Prairie pothole region. Soils diagnostic of dominant water sources.[g]	Aerial photos show variable year-to-year extent of flooding. Hydrographs of water table confirm both short- and long-term fluctuations.	Landscape a mosaic of ponds varying in depth at a single point in time. Floodwaters retained by depressions.	Flyway and breeding sites for waterfowl. Retention of water results in aquatic/moist habitat in otherwise semiarid conditions.
Drawdowns of water table interspersed between frequent rain events that fully saturate sediments.	High evapotranspiration, but supply by rain in poorly drained landscape of impervious sediments creates wetland conditions.	Hydrographs confirm that water table fluctuates widely.	Precipitation and evapotranspiration dominate site-water balance. Floodwaters retained by depressions.	Fluctuating water table conducive to rapid biogeochemical cycling; strong atmospheric exchanges.
Drawdown extreme during course of growing season; periods of flooding brief.	Arid climate or source of recharge minimal.	Hydrographs confirm that water table is low during long periods.	Frequent deficits in site water balance result in ephemeral aquatic ecosystems because of temporary floodwater storage.	Support rare plant and aquatic communities such as vernal pools.[h]
Alternating recharge and discharge varying with stream stage.[i]	Wetland narrow and adjacent to channel; sediment texture coarse.	Water table hydrograph proportional to and coincident with stream hydrograph.	High exchange between channel and groundwater. Temporarily stores floodwaters; conserves/reduces groundwater discharge.	Substrate well aerated and flushed; hydrophytic vegetation may occur in well-aerated soils.
Nearly constant water table at or near surface.	Relatively constant water table position suggests low evapotranspiration because of cool, moist climate; if evapotranspiration is high, strong groundwater discharge required.	Water table hydrographs have little fluctuation and are at or near surface. A cool, moist climate may suggest low evapotranspiration; otherwise, strong groundwater discharge must be assumed.	Stable water table encourages peat accumulation. Where evapotranspiration is low, ombrotrophic conditions promote bog formation. Strong groundwater sources encourage development of fens or maintain seepage slopes.	Landscape patterns dominated by biogenic process of peat accumulation that is vulnerable to changes in drainage and climate. For seepage slopes, species composition reflects waterlogged soils that are nevertheless well flushed and not strongly reduced.
Unidirectional Flow				
Flow velocities correspond with high-gradient landforms.	Coarse sands and cobble sediments; pool and riffle bedform. Evidence of flooding is transient (e.g., debris, lines, tree damage).	Currents strong enough to export fines; particle size confirm high fluvial energy.	Strong currents ensure active geomorphic landform. Wetland well flushed because of high turnover rate of water.	Strong downstream transport processes. Well-aerated water supports coldwater fish populations.

TABLE 5D-4. (cont.)

Examples of Hydrodynamics[a]	Qualitative Evidence[b]	Quantitative Evidence[c]	Functions[d]	Ecological Significance[e]
Unidirectional Flow (cont.)				
Flow velocities correspond with middle-gradient landforms.	Fine to coarse sediments (silts and sands); easily observable flow; point bars develop. Evidence of flooding is persistent (e.g., sediments, disturbance dependent plant communities).	Measurements of flow velocities and sediment particle size confirm middle-gradient condition.	Interspersion between well-flushed and stagnant areas on floodplain. During flooding, strong transport of particulate matter occurs.	Interspersion of low- and high-energy environments supports complex food webs. Possesses capacity to import nutrients or export toxins.
Flow velocities correspond with low-gradient landforms.	Fine sediments (silt-clay and high-organic content); barely perceptible flow during flooding.	Slope, flows, and particle size distribution all confirm low-energy system.	Residence time of water allows long contact between water and sediment; low-suspended load allows light penetration.	Good conditions for trapping sediment and altering water quality. As nutrient trap, food web support is strong. Reducing conditions favor strongly obligate wetland species.
Bidirectional Flow				
Astronomic tides:				
Regular flooding (low marsh or fringe mangrove).	Adjacent to estuary; frequent flooding.	Sea-level controlled; daily to twice-daily tides.	Very active region for biogeochemical process and estuarine food web support.	Many known attributes of intertidal wetlands.
Irregular flooding (high marsh or basin mangroves).	Landscape position landward of low marsh or fringe mangroves.	Flooding during extreme tides and during storms.	Infrequent events transport salt and maintain distinctive ecotone between halophytes and upland.	Leading edge of landward wetland migration responding to rising sea level. Transition establishes barrier to saline water and unique habitat.
Wind-generated water level fluctuations from seiches of large lakes.	Wetland is adjacent to lake:		Shallow water and vegetation contribute to habitat complexity and food production.	Contributes to food web support and habitat maintenance for aquatic and amphibious species in region of otherwise featureless lake bottom and shoreline.
	Strongly influenced by lake if at lake level.	Hydrographs show strong evidence of coupling between wetland water table and wetland surface.		
	Likely groundwater-supported if on slope above lake level; minimal lake input in most years.	Hydrograph of wetland water table proportional to lake level, but consistently above it (except in extreme ground-water drawdown/drought).		

Footnotes on following page.

TABLE 5D-4. (cont.)

Source: Brinson (1993a)

[a] Hydrodynamics is a measure of the kinetic energy of water that takes into account velocity, turbulence, and rate of change of water table position. Because velocity changes so much temporally, sediment particle size can be useful for integrating hydrodynamics.

[b] Potentially useful indices that require little or no laboratory analysis. Usually observable in field.

[c] Velocities can be measured with sensitive current meters; water table fluctuations can be measured in the wetland with continuous stage recorders or frequently monitored wells without instrumentation.

[d] Processes or conditions that contribute to ecological significance.

[e] Other ecologically significant functions may be present; those provided are only examples.

[f] Not mutually exclusive with horizontal unidirectional and bidirectional flows listed below.

[g] Hubbard (1988).

[h] Zedler (1987).

[i] Stanford and Ward (1988).

Brinson has tried recently to emphasize the need for performing functional assessments using HGM only in the context of reference wetlands (Brinson 1995, 1996). That is, once the functions of a reference wetland are well understood, one can assume that wetlands with similar HGM attributes will function in similar ways (i.e. that ecosystem function can be inferred from ecosystem structure). The efficiency or strength of each function can then be combined into a functional index to determine the relative merits of numerous similarly situated sites. For instance, the functional index with respect to nutrient removal will combine information on inflow rates, outflow rates, and residence time as measured by wetland volume and hydraulic roughness. The index can then be compared across evaluation sites to determine relative efficiencies of nutrient removal. However, comparison with a reference wetland (where something is known about the processes and rates of nutrient removal) is required to bring this relative functional assessment into a more absolute realm. Barring a thorough data set on regional reference wetlands, most wetland practitioners are faced with using HGM in very relative terms and combining their findings with professional judgment to perform functional assessments.

In addition, HGM cannot be directly linked to Cowardin et al. (1979) without some level of regional knowledge about wetland ecosystems. Although there are a finite number of plant community types within each HGM classification, there are few HGM types with a uniquely determined plant community type (e.g., even an ombotrophic bog can have forested, scrub-shrub, and moss/lichen zones or patches).

Given the caveat that local natural history is necessary to perform functional assessments using HGM (or to link HGM and Cowardin et al. [1979]), substantial information regarding the structure and function of specific ecosystems can be obtained from limited site-specific observations using these tools. For example, peatland systems require a positive water balance; variation in the duration of soil saturation (associated with topographic variation) largely controls the extent of shrub or forest cover. Bottomland hardwood forests are located within the floodplains of low-gradient alluvial systems where water comes from lateral surface or near-surface transport from overbank flows and where flood hydrographs are low and broad. The specific mix of tree species is again highly dependent on the immediate variation in the duration of soil saturation. Patterns of soil moisture are related to the topographic variation associated with natural levees, terraces, or tributary and overflow channels. Also associated with topographic variation are differences in the deposition rate, nutrient flux, and grain size of deposited materials.

Water Regimes Used in Cowardin et al. (1979)

To fully describe a wetland using the classification of Cowardin et al. (1979), water regime modifiers are used. Given that the required data on site hydrology are often not available, an understanding of site hydroperiod (as a function of geomorphic position and water source) developed under HGM is necessary to determine the appropriate water regime modifier. These modifiers further help to provide the link between HGM, the plant communities that one encounters, and their associated functions. Water regimes are first divided into two major categories, tidal and non-tidal, of which only the non-tidal category is used in freshwater settings. Tidally influenced portions of freshwater wetlands (as described in the previous section (*Estuarine and Coastal Wetlands*) are described using the appropriate nomenclature with an appended "-tidal" if the nomenclature is derived from the non-tidal category.

Even if not influenced by oceanic tides, non-tidal water regimes may be affected by wind or seiches in lakes. These areas are often referred to as freshwater estuaries and are more common at the mouths of low-gradient rivers draining the Great Lakes. Water regimes are defined in terms of the growing season, which are equated to the frost-free period (see the U.S. Department of Interior National Atlas 1970:110-111 for a generalized regional delineation). The rest of the year is defined as the dormant season, a time when even extended periods of flooding may have little influence on the development of plant communities. Non-tidal modifiers (Cowardin et al. 1979) are as follows:

- **Permanently Flooded**—Water covers the land surface throughout the year in all years and vegetation is composed of obligate hydrophytes.
- **Intermittently Exposed**—Surface water is present throughout the year except in years of extreme drought.
- **Semipermanently Flooded**—Surface water persists throughout the growing season in most years. When surface water is absent, the water table is often near the land surface.
- **Seasonally Flooded**—Surface water is present for extended periods, especially early in the growing season, but is absent by the end of the season in most years. When surface water is absent, the water table is often near the land surface.
- **Saturated**—The substrate is saturated to the surface for extended periods during the growing season, but surface water is seldom present.
- **Temporarily Flooded**—Surface water is present for brief periods during the growing season, but the water table usually lies well below the soil surface for most of the season. Plants that grow

both in upland and wetland areas are characteristic of the temporarily flooded regime.

- **Intermittently Flooded**—The substrate is usually exposed, but surface water is present for variable periods without detectable seasonal periodicity. Weeks, months, or even years may intervene between periods of inundation. The dominant plant communities under this regime may change as soil moisture conditions change. Some areas exhibiting this regime do not fall within any commonly accepted definition of wetlands because they do not have hydric soils or support hydrophytes.
- **Artificially Flooded**—The amount and duration of flooding is controlled by means of pumps or siphons in conjunction with levees, dikes, or dams. The vegetation growing on these areas cannot be considered a reliable indicator of water regime. Examples of artificially flooded wetlands are some agricultural lands managed under a rice-soybean rotation and wildlife management areas where forests, crops, or pioneer plants may be flooded or dewatered to attract wetland wildlife. Neither wetlands within or resulting from leakage from man-made impoundments nor irrigated pasturelands supplied by diversion ditches or artesian wells are included in this modifier.

ECOSYSTEM PROFILE

Because so much of what has been learned about freshwater wetlands has been learned in a regulatory environment, numerous management classification schemes exist. Cowardin et al. (1979) is the most well established, forming the basis of the National Wetland Inventory maps, which are a primary source of wetland data. Cowardin et al. (1979) is based on dominant vegetation type and is, therefore, independent of the HGM classification described above. Because of its widespread acceptance, wetland classifications, as described in Cowardin et al. (1979), will be the basis for the following discussion of freshwater (i.e., not riverine or lacustrine) wetlands. Many freshwater wetlands contain patches (or rings) of several wetland classes (Figure 5D-3) related to the flooding regime within the specific patch or zone. Sites that are the purest examples of their class are described below. In a single wetland, these classes may shift from one into another across time (temporal heterogeneity and succession) and space (spatial heterogeneity), giving rise to wide local variability in vegetation and process. Freshwater wetlands may contain patches of several wetland classes within a contiguous wetland complex. Furthermore, freshwater wetlands often develop adjacent to rivers and lakes. Riverine and lacustrine ecosystem processes and characteristics can have a significant impact on freshwater wetlands in these situations.